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Towards a Sustainable Management of the Food Chain
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Organised by Agroscope Reckenholz-Tänikon Research Station ART

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Conference Organisers

Agroscope Reckenholz-Tänikon Research Station ART
Reckenholzstrasse 191, CH-8046 Zürich, Switzerland

Email:	lcafood08@art.admin.ch
Conference website:	www.lcafood08.ch
ART website:	www.art.admin.ch
Phone:	+41 44 377 71 72
Fax:	+41 44 377 72 01

Scientific Committee

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Editors

Thomas Nemecek and Gérard Gaillard

Layout

Ursula Kläger

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Editorial

Agriculture and the food sector are responsible for a large share of the environmental impacts and resource use caused by human activity. For certain environmental issues such as the conservation of biodiversity, agriculture is the key driver. For about 15 years now, the Life Cycle Assessment (LCA) method has successfully been used to analyse agricultural production systems and food chains. During the five previous conferences held in Belgium, Sweden and Denmark, the scientific community discussed LCA topics in the Agri-Food Sector. The 6th International Conference on LCA in the Agri-Food Sector was organised in Zurich on 12-14 November 2008, with the following objectives:

- to show recent developments in terms of methodology, approaches, databases and tools;
- to present applications of the LCA methodology in new case studies or case studies showing new aspects in various food chains;
- to present successful examples of communication of LCA results to stakeholders and their use in decision making.

The conference has received a much higher attention than the 5th conference held in April 2007. The number of participant rose from 61 to 160, the submitted abstracts from 60 to 150. A total of 51 oral presentations were held during twelve sessions, compared to 27 presentations during the previous conference. The participants presented also 62 posters. These figures illustrate the growing interest and the increasing activities in the field of LCA in the agri-food sector. The participants originated from 32 countries, with an increasing proportion of participants from outside Europe, particularly from non-OECD countries (Fig. 1). Still, three quarter of the participants came from European countries. We were happy to see several new organisations starting work on LCA in the agri-food sector.

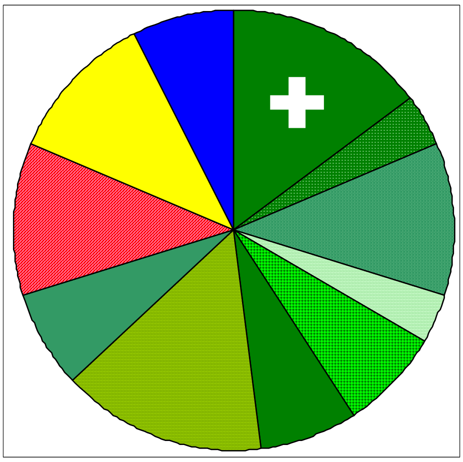
An increasing activity was observed in the following fields: databases and tools, assessment of land and water use, ecotoxicity, food processing, decision support and linking to economic assessments (Fig. 2). The contributions from emerging countries were increasing, but still scarce. Life cycle and food chain management received more attention than before. There was also an evolution from isolated case studies with limited representativity to a wider scope on sectoral, national or supra-national level (like the EU-27). Methodical progresses have been made in assessing impacts specific to agriculture, like land use, biodiversity and water resources. Several contributions extended the classical environmental LCA to a full sustainability analysis. Some progress has been made on regionalisation of LCA, but a lot of work still lies before us. Progress has also been made on databases and tools.

For the future LCA research, we see among others the following key issues:

- Considerable efforts should be invested in the improvement of the methodology. In particular standard and widely recognised methods for the assessment of land use, water resources and pesticide impacts are still missing, which limits the validity of the results. Pharmaceuticals are ignored in almost all LCAs.
- Despite the fast computers and adapted software, we see still very little assessments of the variability and uncertainty.
- We should not forget that communication to decision makers, stakeholders and the public is a key issue, not only for ensuring funding. The decision makers need not be familiar with the details of the methodology, but they have to understand the results and conclusions and they must be convinced that the recommendations given are the way forward.
- Last but not least, LCA applications in non-European and particularly non-OECD countries should be promoted. The potential to make the agri-food sector more environmentally friendly in these countries is much bigger than in the European countries, where LCA had its origin. Furthermore, the food consumed in the industrialised countries has its origin in all continents.

These proceedings give the full papers of the oral presentations during the conference. All manuscripts have been peer-reviewed by members of the scientific committee. We would like to express our thanks to the scientific committee for its big effort and the local organising committee for the smooth organisation of the conference. We are looking forward to the 7th conference in Bari on 22-24 September 2010 (www.lcafood2010.uniba.it)

Zurich, June 2009 Gérard Gaillard and Thomas Nemecek



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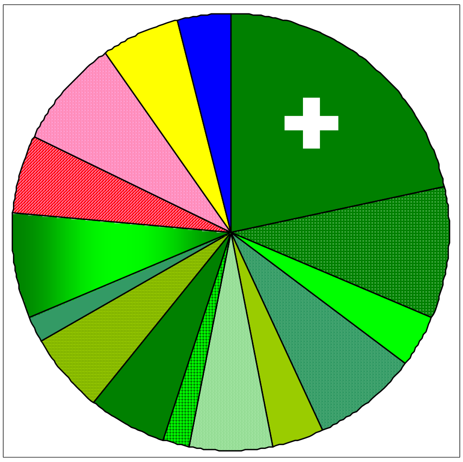

LCAfood 2008

Fig. 1: Comparison of the countries of origin of the LCA Food conference 2007 in Gothenburg and the LCA Food conference 2008 in Zurich.

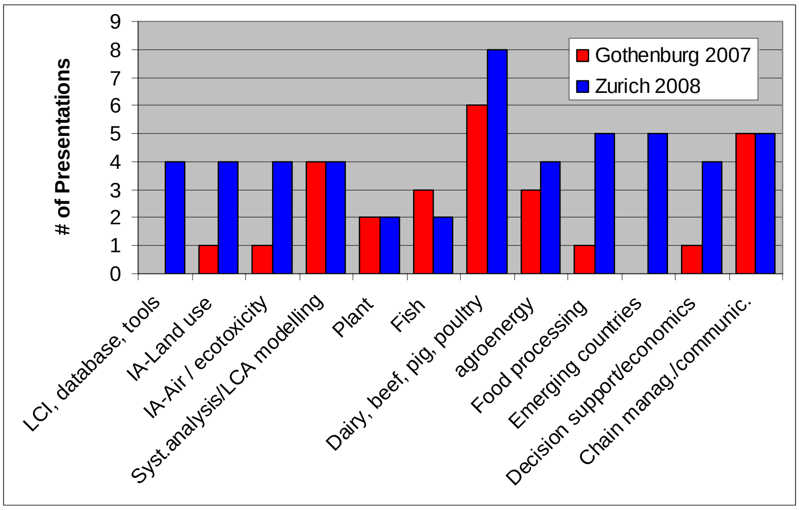

Fig. 2: Distribution of the presentation during the LCA Food conference 2007 in Gothenburg and the LCA Food conference 2008 in Zurich

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Assessing freshwater use impacts in LCA

L. Milà i Canals[1,2], J. Chenoweth[1], A. Chapagain[3], S. Orr[3], A. Antón[4] and R. Clift[1]
[1] Centre for Environmental Strategy, University of Surrey, GU2 7XH Guildford (Surrey), UK
[2] Current address: Unilever – Safety & Environmental Assurance Centre, Colworth Park, Sharnbrook, Bedfordshire, MK44 1LQ, UK (Llorenc.Mila-i-Canals@unilever.com)
[3] WWF-UK, Panda House, Weyside Park, GU7 1XR Godalming (Surrey), UK
[4] IRTA, ctra. Cabrils, km 2, 08348 Cabrils (Barcelona), Spain

Keywords: water footprint; water resource; freshwater ecosystem impact; LCA; evaporative use; ecosystem; freshwater depletion

Abstract

This presentation describes the main impact pathways related to changes in the amount of water available for ecosystems and future generations (i.e. qualitative aspects are not included). Freshwater flows requiring distinction in the LCI are discussed and quantified, including evaporative and non-evaporative uses of blue and green water, and land uses leading to changes in freshwater availability. Suitable indicators are suggested for the two main impact pathways (namely freshwater ecosystem impact, FEI, and freshwater depletion, FD) and operational characterisation factors are applied in the studied countries. For FEI, an indicator relating current freshwater use to the available freshwater resources is suggested. For FD, the parameters required for the implementation of the commonly used Abiotic Depletion Potentials are explored and illustrated.

Applying this framework in a case study of broccoli production in the UK and Spain for consumption in the UK serves to discuss advantages and potential drawbacks for its widespread use. This methodological framework improves the representation of freshwater use derived impacts in LCA.

Introduction

As discussed by Milà i Canals *et al*. (2009), water is a precious and increasingly scarce resource. It is critical for ecosystem functions (as both habitat and resource) and equally essential for humans. Water abstracted for human purposes can have significant impacts on water systems. Over 100,000 species (almost 6% of all described species) live in fresh water and countless others depend on fresh water for survival (Dudgeon *et al*. 2005). Freshwater species and habitats are more imperilled globally than their terrestrial or marine counterparts (WWF 2006). In the most extreme cases, water scarcity has resulted in complete ecosystem collapse (Micklin 1988). Similarly, some major rivers have periodically completely dried up, including the Rio Grande/Bravo in Mexico and the Great Ruaha River in Tanzania (WWF 2007).

In contrast with this, water use impacts have been underrepresented since the start of LCA methodology in the late 1960s, probably due to LCA being developed for industrial systems (usually less dependent on water resources than agricultural ones) in water-abundant countries. Basically, LCA studies report the total amount of water used by the production system, from cradle (raw material acquisition) to grave (waste management). In general, such studies do not even distinguish the source from which water is obtained or the way or condition in which water leaves the product system.

Outside of the field of LCA, the concept of Virtual water (VW) has evolved since the early 1990s and refers to the amount of water required to produce a certain product (Allan 1998, 2001). VW studies have taken on more precise and practical applications since Hoekstra & Hung (2002), Chapagain & Hoekstra (2003, 2004), Chapagain & Orr (2009; 2008), began to quantify and calculate VW flows and related water footprints (WF). Today the concept of WF is gaining momentum within industries, and some expect it to be as successful as carbon footprints.

This contribution explores links between the WF methodology and LCA, and suggests ways to represent the impacts related to water use in the life cycle impact assessment (LCIA) phase.

Methods

System boundaries and studied systems

Water use related to the production, distribution and consumption of broccoli in the UK has been studied from a cradle to grave perspective, up to the point of digestion and excretion of human waste (Muñoz *et al*. 2008; Milà i Canals *et al*. 2008). The studied systems include production of broccoli in the UK for fresh consumption from April to November; production in the UK and freezing for consumption from November to April; and Spanish production and distribution to the UK for fresh consumption from November to April. An extensive description of the studied supply chains is offered in Milà i Canals *et al.* (2008). The supply chains are coded according to the country of origin (ES or UK); farm number (1 and 2 in Spain; 5 and 6 in the UK); a 1 or a 2 for early or late crops; and the suffix "fr" for frozen supply.

Impact pathways considered

As thoroughly discussed by Milà i Canals *et al*. (2009), the following four main impact pathways related to freshwater use may be distinguished and merit attention in LCA; they are illustrated in Fig. 1:

1. Direct water use leading to changes in fresh water availability for humans leading to changes in human health;
2. Direct water use leading to changes in fresh water availability for ecosystems leading to effects on ecosystem quality (Freshwater Ecosystem Impact, FEI);
3. Direct groundwater use causing reduced long-term (fund and stock) fresh water availability (Freshwater Depletion, FD);
4. Land use changes leading to changes in the water cycle (infiltration and runoff) leading to changes in fresh water availability for ecosystems leading to effects on ecosystem quality (FEI).

Only the impacts on ecosystems' quality (from direct water use and from land use) and on freshwater depletion are further considered in this contribution.

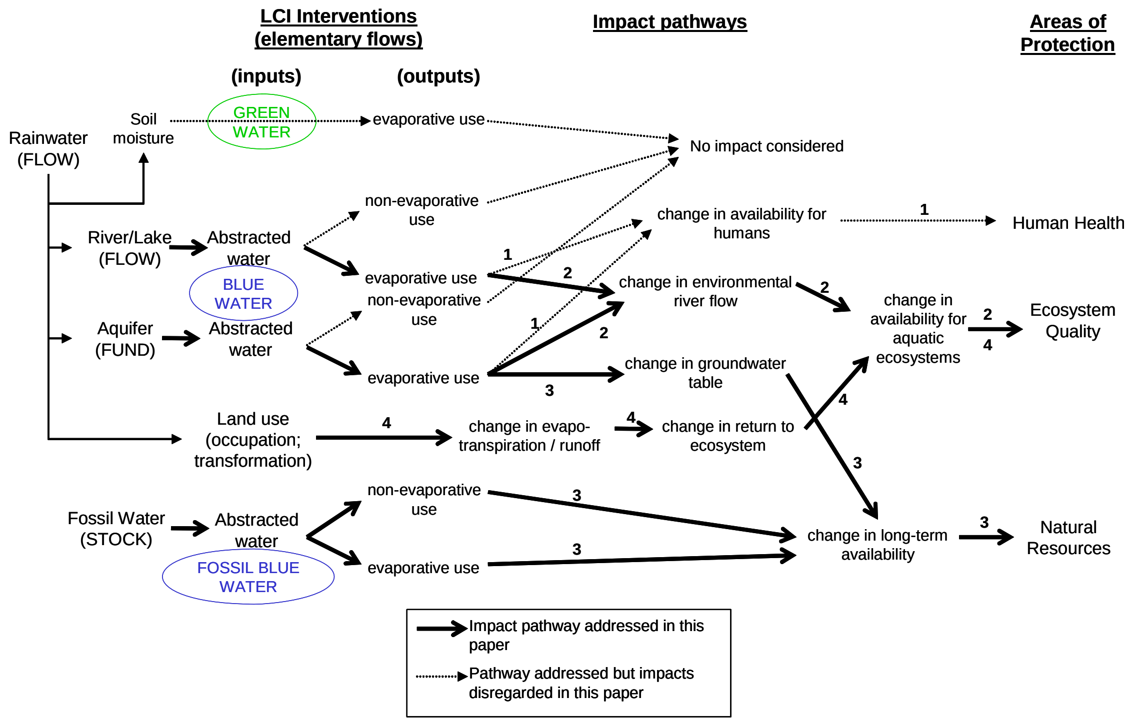

Fig. 1: Main impact pathways related to freshwater use. Only those pathways depicted with solid arrows are considered for LCA. The concepts in circles denote common denominations in the Water Footprint field. The numbers refer to the impact pathways defined in sections 3.1–3.4 of Milà i Canals *et al.* (2009).

LCI: quantification of environmental interventions

Guidance on how to calculate water use flows is offered in Milà i Canals *et al.* (2009). This focuses on abstracted (blue) water to be used in human activities. Water occurs in the form of *green water* (stored as soil moisture and available for evaporation through crops and terrestrial vegetation) and *blue water* (surface or groundwater). Blue water is the volume of water in ground (aquifer) and surface water bodies available for abstraction. The distinction between blue water and green water is important as green water is only available for use by plants at the precise location where it occurs, whereas blue water is available generally for use in a wide range of human managed systems, including but not limited to use by plants. WF calculations (e.g. Chapagain & Orr 2009; 2008) generally distinguish the two types of water, but in LCA we recommend to use the WF approach to calculate water flows but focusing on blue water, as this is the one that can be linked to impacts on ecosystems and depletion (Milà i Canals *et al.* 2008). In addition, land use and land use change may be linked to changes in water availability for ecosystems due to differences in evaporative use respect a reference system; for "sealed"-type land uses, also runoff water is considered to be lost for ecosystem use (Milà i Canals *et al.* 2009).

Milà i Canals *et al.* (forthcoming) illustrate how to assess the fraction of water evaporated in different life cycle uses, from the volumes and land use interventions identified through LCA databases (such as ecoinvent).

LCIA: characterisation factors for FEI and FD

Milà i Canals *et al.* (2009) provide characterisation factors (CF) for the Freshwater Ecosystem Impact (FEI) using two different indicators: Water Use Per Resource (WUPR) and Water Stress Indicator (WSI). Here only the WUPR for the relevant countries is used (Spain: 32%; UK: 6.5%). In addition, in many background processes the only geographical reference is "Europe"; therefore, a new CF had to be derived for Europe in terms of WUPR, from total use of water and total water resources in Europe. The WUPR for Europe is 15% (Milà i Canals *et al.* forthcoming). This characterisation factor has also been used for all water flows where origin is not specified. The Swiss Eco Scarcity Method 2006 also uses WUPR as an indicator for freshwater use impacts (Frischknecht 2008).

In the case of FD, Custodio (2002) points out that the aquifer in Murcia is reportedly overexploited (depletion rate of $125x10^6m^3yr^{-1}$ on remaining reserves of $10,000x10^6m^3$, data for 1995). Therefore, the formula for the Abiotic Depletion Potentials (ADP) suggested in Milà i Canals *et al*. (2009) is used to derive an ADP for all groundwater uses in the Spanish crops as $1.77x10^8$kg Sb-eq/kg (Milà i Canals *et al*. forthcoming).

Results

Fig. 2 (left) shows LCI results for water use on a cradle-to-grave analysis of broccoli using a "typical LCA approach", i.e. a quantification of blue water use. In these results all uses of abstracted water are shown per life cycle stage from cropping through to processing (packing; freezing), transport and retail, home storage and cooking, and (consumption and) excretion of food. When the crops are irrigated (in the Spanish –ES- systems) the cropping stage dominates the results, although contributions from the background system are notable. The latter appear mainly in the 'Home' stage and are mostly related to electricity consumption, as well as in the 'Excretion' stage, where they arise mainly through toilet use (Muñoz *et al*. 2008; Milà i Canals *et al*. 2008). In Fig. 2 (right) the most relevant flows from a freshwater ecosystems perspective, i.e. evaporative use of blue water and water rendered unavailable for ecosystems through land use (Milà i Canals *et al*. 2009), have been highlighted in the solid (blue and brown) columns. Additionally, evaporative green (rain)water use and non-evaporative blue water use are shown for information.

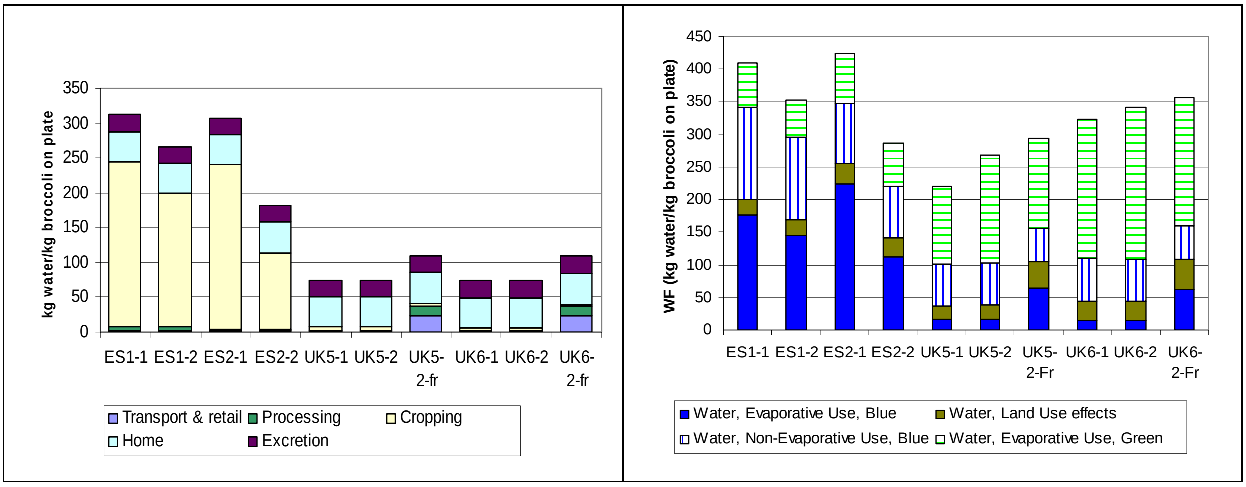

Fig. 2: Left: Total (blue) water use in the life cycle of broccoli (Milà i Canals *et al*. 2008). Right: 'WF-LCA' results for water use: Evaporative blue water use (solid blue columns); land use effects on water availability (brown columns); evaporative green water use (horizontal stripes); and non-evaporative blue water use (vertical stripes) in the life cycle of broccoli (Milà i Canals *et al*. forthcoming).

Evaporative green water use (in horizontal stripes) and land use effects (brown) are the main differences respect Fig. 2 (left). This shows that from a total water consumption point of view the differences between British and Spanish crops are not so big; however, the environmental relevance of consuming rainwater is minor (Milà i Canals *et al*. 2009). The non-evaporative blue water use (vertical stripes) would only be relevant from an abiotic resources depletion potential point of view in the cases where water was abstracted from overexploited aquifers (Milà i Canals *et al*. 2009), as is partly the case in the Spanish crops assessed (Murcia). Most (50-70%) of the WF shown in Fig. 2 (right) is caused by the cropping stage, i.e. it is water evaporated by the crop or lost as runoff / leak. This result was expected as agriculture is the main water user. However, it is interesting to note the other main sources of water use identified here: land use effects on the water balance (6-14% of water use) and electricity use (15-50%; used for cooking, refrigeration, irrigation, etc.), followed by other minor contributions.

ES1 has relatively higher non-evaporative use of water than ES2, due to the more inefficient irrigation system: ES1 uses gravity irrigation, where only 70% of water has been assumed to be available for evaporation by the crop as opposed to ES2 where drip irrigation, with 90% efficiency, is in place. In the case of production sites in the UK, plenty of rainfall is available during the growth period of broccoli which meets a large part of the evaporative demand of the crop, minimising the need for irrigation water use. In practice, this crop is usually not irrigated at all, because broccoli may stand some water stress better than other more delicate crops (such as lettuce, which also needs irrigation in the UK). There are two reasons why the green virtual water content of the broccoli from UK5-1 is smaller than that from the UK6-1. The main reason is that the crop yield per unit of land is relatively low in UK6 (15,600 kg per hectare of land in UK5 compared to the 9,600 kg per hectare in UK6). Hence, with a similar magnitude of evaporation, the crops in UK6-1 evaporate more water per kilogram of crop. The second and minor reason is that the planting at UK5-1 starts in mid March whereas in UK6-1 it starts early April. This makes the effective rainfall available in the first site smaller than the second one (the first site effectively uses 144 mm from rainfall whereas the second one evaporates 158 mm per season of the crop).

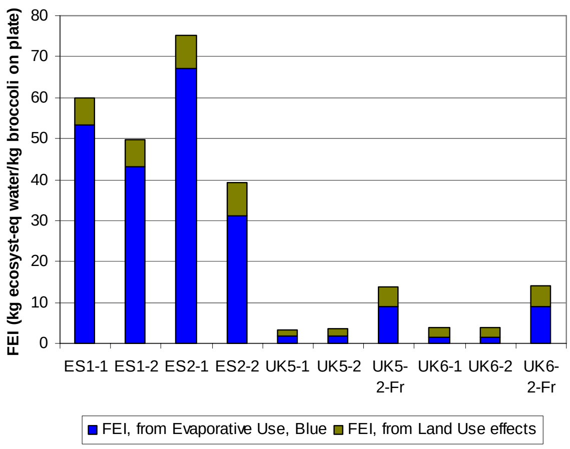

Fig. 3: Characterised results of Freshwater Ecosystem impact (FEI), in litres of “ecosystem equivalent water” per kg broccoli on plate: distinction is made between impacts from direct evaporative blue water use (blue columns) and land use derived impact on freshwater availability for ecosystems (brown columns) in the life cycle of broccoli (Milà i Canals *et al*. forthcoming).

In terms of impact assessment, Fig. 3 shows the results for the newly defined impact category “Freshwater Ecosystem Impact” (Milà i Canals *et al*. 2009). The indicator is defined as volume or mass of “ecosystem-equivalent” water, referring to the volume of water likely to be affecting freshwater ecosystems (Milà i Canals *et al*. 2009). Applying the characterisation factors exaggerates the differences between Spanish and British systems already shown in Fig. 2 (left). This is due to several reasons: first, green water use (main water use in the UK, see Fig. 2 right) has a zero impact. In addition, the characterisation factor is 0.32 for Spain and 0.065 for the UK (i.e. 32% of water resources are being used in Spain, while only 6.5% are being used in the UK); thus, each m^3 of water used in Spain is regarded as having a higher impact than the equivalent amount in the UK. Because most of the water is used in the cropping stage (Fig 2 left) in Spain, the different characterisation factors have a profound effect on the results. The effect from land use is again relevant, but does not dominate the results. Due to the differences in annual rainfall the land use effects are also more pronounced in Spain (Milà i Canals *et al*. forthcoming).

In the case of Depletion of Abiotic Resources (results not shown), once the use of water from Murcia’s over abstracted aquifer is considered in the Spanish farms it completely dominates the results. In fact, it causes the contributions to this impact by the Spanish farms to be twelve orders of magnitude above the contributions from British farms, which are dominated by energy resources (oil, gas, etc.) (Milà i Canals *et al*. forthcoming).

Discussion

When only the evaporative use of water is included in the impact assessment, farms using water more inefficiently and effectively abstracting more water (such as ES1 compared to ES2) seem to cause a smaller impact on freshwater ecosystems (Fig. 3). This is rather counterintuitive, and as discussed by Milà i Canals *et al*. (2009) a more precautionary approach would suggest including total abstraction (evaporative + non-evaporative use) in this impact category. So far, applying LCIA characterisation factors to the relevant volumes identified in the LCI does not seem to cause much difference in the results. However, in cases where irrigation water is also used in a water-abundant country the comparison would change dramatically between LCI and LCIA results. For instance, Hospido *et al*. (submitted) report similar (blue) water uses in UK- and Spain-grown lettuce; applying the LCIA approach suggested here would probably show that in terms of potential impact on freshwater ecosystems, water use in Spain is much more significant.

Recent moves towards a taxation system for groundwater use in Southern Spain might radically change the usage patterns of overexploited aquifers, which would in turn change the calculated ADP and potentially affect the results for Freshwater Depletion commented here.

Conclusion

This methodological framework improves the representation of freshwater use derived impacts in LCA. The method should be tested with further case studies in order to decide suitability and necessity of the LCIA characterisation factors. In particular, the current case has obvious results because it compares an irrigated crop in a water scarce region with a rain fed crop in a water abundant country. This study has identified other major sources of water use, besides agriculture, in the life cycle of vegetables, namely direct water use for cooking and sanitation, land use effects on the water cycle and electricity production.

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Regionalised LCIA of vegetable and fruit production: Quantifying the environmental impacts of freshwater use

S. Pfister[1], F. Stoessel[1], R. Juraske[1] , A. Koehler[1] and S. Hellweg[1]
[1]ETH Zurich, Institute of Environmental Engineering, ETH Hönggerberg, 8093 Zurich, Switzerland, pfister@ifu.baug.ethz.ch

Keywords: LCIA, vegetables, freshwater use, regionalisation, Eco-indicator 99

Abstract

Many LCA studies of agricultural products neglect the impacts of water use. In this paper we provide a regionalised study based on new inventory data including water use figures for the following agricultural products: tomatoes, potatoes, cabbage, onions, and peppers. We developed and applied a method to assess the environmental damages resulting from freshwater use. Our method is concordant with the framework of the Eco-Indicator 99 method (EI99) and allows the integration into standard LCA studies to show the relevance of water use related environmental impacts.

Because environmental consequences of water use are of high spatial variability, the assessment was performed with inventory data for the agricultural production in seven different countries with different climatic and socio-economic conditions: Switzerland, Spain, China, Greece, Italy, USA, and Ethiopia. Region-specific impact factors were developed and applied. The results show that in some countries environmental impacts due to water use can be relevant or even dominate the environmental damages of agricultural production. We also compared water use with land use impacts which are significant when applying the standard LCA methodology on field-grown vegetables.

The results of this work highlight the importance of integrating water use in LCA studies on agricultural products and pinpoint the relevance of regionalisation on the level of the inventory analysis as well as impact assessment.

Introduction

Agricultural production is one of the most important economic activities and responsible for about 70% of the global anthropogenic freshwater withdrawals, while only 20% and 10% are used by industry and the municipalities, respectively (WB 2004). Furthermore, freshwater scarcity has been recognized as one of the most crucial environmental issues (UNESCO 2006) and several regions around the world are already facing this problem. Yet, environmental impacts caused by freshwater use are generally not considered in LCA studies.

Attempts to integrate water resources into life cycle assessment have been limited to conceptual contributions (e.g. Owens 2002) and simplified methods, such as the cumulative exergy demand (CExD) (Bösch *et al.*, 2007), which does not account for regional differences in ecological impacts related to water use. Such regional aspects, however, are very relevant (Vörösmarty *et al.* 2005), especially for products with a globalized value chain. The distance-to-target method of Ecological Scarcity 2006 accounts for regional aspects by assessing freshwater use on country level (Frischknecht *et al.*, 2008). National water-stress values are used to derive impact factors based on a defined threshold. While this method is a good first step to quantitatively assess potential water stress, it does not differentiate between water consumption (e.g. evaporation) and other water use (e.g. use of water as a cooling agent, returning the water to the watershed after use). Recently, another methodological framework was proposed by Mila i Canals *et al.* (2008), providing midpoint characterization factors based on water use-to-availability ratios. However, these midpoint factors cannot be applied for assessing the relevance of water use compared to total impacts of crop production in LCA.

In addition to the lack of appropriate LCIA methods, no generally accepted standards for water-use reporting exist in LCA and adequate inventory schemes are missing (Koehler 2008). Particularly, for agricultural production regionalized inventories are crucial as agricultural freshwater use is very dependent on the climate as shown by tools such as CROPWAT (FAO, 1999).

Virtual water data are one available data source which report water requirements for several crops and countries (Chapagain & Hoekstra, 2004). However, these data cannot directly be used as, in LCA, we need to quantify irrigation water and not total crop water requirements.

This paper presents the relative impact of water use in regionalized LCA of vegetables based on new inventory data and a new LCIA method of water use. It explores the importance of water use in relation to total impact and impact from land. Furthermore, the relevance of regionalization in LCA of agricultural products is presented.

Approach

Regionalized Inventory

Stoessel & Hellweg (2008) developed new life cycle inventory (LCI) data for vegetables and fruits produced in different countries. These LCI data sets include also irrigation water requirements. The water requirements are calculated based on national virtual water data (Chapagain & Hoekstra, 2004) and regional precipitation data.

Impact assessment of water use

We applied the method developed by Pfister *et al.* (submitted) to assess the environmental impacts of freshwater use in vegetable production. This method is designed to complement the Eco-indicator 99 (EI99) method (Goedkoop & Spriensma, 2001) by modeling the impact pathways of damages to three areas of protection (AoP) human health, ecosystem quality, and resources. This approach is also consistent with the framework proposed by the UNEP-SETAC Life Cycle Initiative project *Assessment of use and depletion of water resources within the LCA Framework (WULCA)* (Bayart *et al.*, submitted).

As direct environmental impacts from polluting water are generally considered by impact assessment methods for emissions, the method of Pfister *et al.* (submitted) assesses only damages caused by water consumption which, in the case of agricultural production, mainly represents the evaporation. The method does not model any ecological impact arising from the water used and released back to the watershed. This type of freshwater use is considered as degradative water use because of the deteriorated quality.

Unlike other abiotic resources, freshwater is indispensable to life and consequently has a crucial role for ecosystem quality and human health. Furthermore, water exists both, as a renewable flow resource (same as e.g. sunlight) but also as funds or deposits (e.g. fossil groundwater). Flow resources have so far not been addressed in the EI99 but conceptually described in the CML method (Guinée, 2001).

Consumption of freshwater deposits or overuse of stocks can be assessed according to the AoP “resources” attributing, for instance, surplus energy [MJ] to the unit of water consumed for accounting for the impact on future users. Surplus energy is the additional amount of energy required by a potential backup technology to provide the resource in future. Pfister *et al.* (submitted) used as ultimate backup technology desalination of seawater. On the other hand, consumption of renewable water resources, particularly freshwater flows, may lead to direct impact on human health and ecosystem as competition will lead to reduced water availability for some users.

Pfister *et al.* (submitted) quantified impacts on the natural environment, which are in general of main importance for water use, combining vulnerability of the vegetation regarding water shortage and the regional water availability. The derived impact factors are measured as potentially damaged fraction during the “occupation” of an area [PDF•m^2•year] per unit of water consumed which is directly comparable to impacts caused by land use.

Finally, damages to human health were assumed to be primarily caused by lack of water for agriculture production and measured in disability adjusted life years lost [DALY] according to this method (Pfister *et al.*, submitted). This impact pathway considers the population vulnerability to lacking freshwater for agricultural production, the resulting health impacts and a water stress index.

As water availability and ecological impacts caused by freshwater use are highly spatially dependent, a regionalized impact assessment is necessary. In the method, we used two levels of regionalisation in order to allow impact assessment on both, the country level (due to higher data availability) and on the watershed level to better represent hydrological features. The spatial differentiation is demonstrated in Fig. 1 for the case of Europe.

Global data of annual hydrological water availability and anthropogenic water use, which are the basis for calculating water stress and overuse of water resources, are provided by the WaterGAP2 model (Alcamo *et al*., 2003). We enhanced this data by integrating the effect of seasonal and inter-annual variability of precipitation using data of Mitchell & Jones (2005).

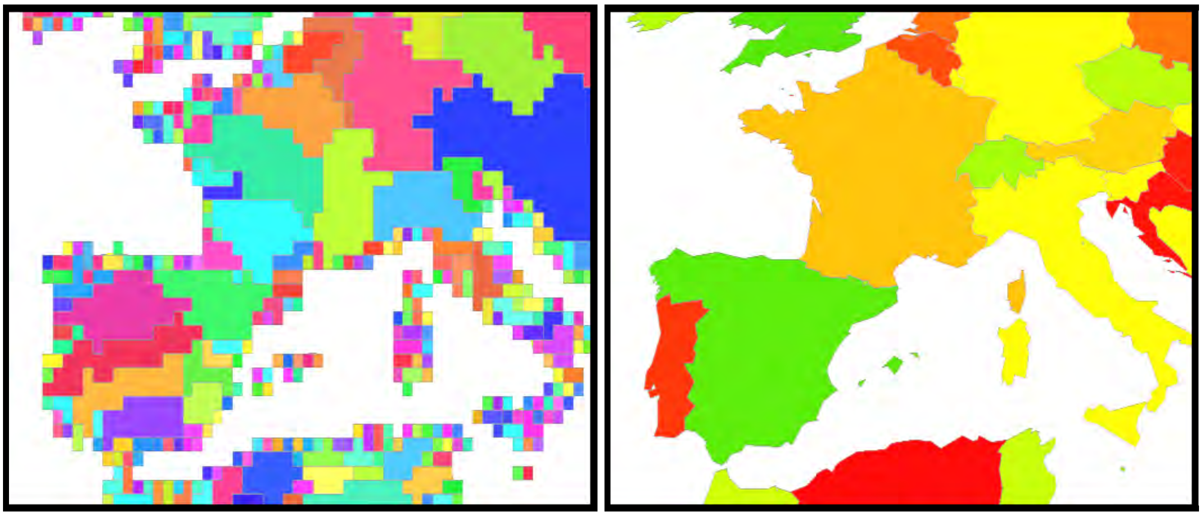

Fig. 1: Maps showing spatial units for the two level of regionalization. Left: watersheds as provided in Alcamo *et al*. (2003). Right: countries. Note that watersheds are not always smaller than countries (e.g. Switzerland is smaller than its watersheds Rhine, Rhone, Danube and Po) but they represent hydrological units relevant for impact assessment.

Regionalized LCA of vegetable production

We elaborated a regionalized LCA study of vegetable production for tomatoes, potatoes, cabbage, onions, and peppers in Switzerland, Spain, China, Greece, Italy, USA, and Ethiopia using the LCIA method EI99. In order to assess the relevance of freshwater use, we included the assessment of irrigation water applying our newly developed LCIA methodology. Process water and water used in background processes was neglected as it is, in general, considered negligible compared to irrigation water for agricultural production.

We used water-consumption characterization factors for countries as well as for selected watersheds within the countries (where relevant crops are grown) in order to show the relevance of different regionalization levels, especially for countries with large areas.

In addition, we compared the environmental impacts of water use in relation to impact of land use as these two activities are particularly important in agriculture and similar ecological damages can arise.

Results

The results of the regionalized LCA study on vegetables show that impacts of water use can be insignificant (e.g. in Switzerland) or even dominate the overall results (for onion production in China and pepper production in Spain). These contrary results depend on the irrigation requirements and regional impact factors (Tab. 1). In vegetable production in Spain, China, and the USA water use generally has a relevant environmental impact, whereas mainly Switzerland it has less importance. This result is visible from the assessment on watershed level, while on country level these trends are less obvious (Tab. 1). National averaged impact factors of China, Spain and the USA are far below the factors in the specific watersheds.

The relevance of water use for different vegetables is characterized by large variation. Water use in tomato production is usually not relevant. Compared to land use, freshwater use can be relevant for all vegetables (Tab. 2).

Tab. 1: Impacts due to water use compared to the total LCIA score for different crops in different countries according to Eco-indicator 99. Relative damage of water use in percent of results from total standard LCA is classified as follows: <10% = " -- ", 10-20% = " - ", 20-50% = " +/- ", 50-100% = " + " and >100% = " ++ ". Impact of water use is analysed on watershed level (upper part) and on country level (lower part) showing the relevance of regionalisation

	Switzerland	Spain	China	Greece	Italy	USA	Ethiopia
				watershed level			
Onion	--	+/-	++	--	--	+/-	+/-
Tomato	--	-	+/-	--	--	-	--
Potato	--	+	+	--	--	+	-
Pepper	--	++	-	-	-	+	+/-
Cabbage	--	+/-	+	-	--	+	+/-
				country level			
Onion	--	+/-	+/-	-	--	+/-	-
Tomato	--	--	--	--	--	--	--
Potato	--	+/-	+/-	-	--	+/-	-
Pepper	--	+/-	+	+/-	-	+/-	+/-
Cabbage	--	+/-	+/-	+/-	--	+/-	+/-

Tab. 2: Impacts due to water use compared to the impact of land use for different crops in different countries according to Eco-indicator 99. Relative damage of water use in percent of results from land use is classified as follows: <10% = " -- ", 10-20% = " - ", 20-50% = " +/- ", 50-100% = " + " and >100% = " ++ ". Impact of water use is analysed on watershed level (upper part) and on country level (lower part) showing the relevance of regionalisation.

	Switzerland	Spain	China	Greece	Italy	USA	Ethiopia
				watershed level			
Onion	--	+	++	-	--	+/-	+
Tomato	--	++	++	+/-	+/-	++	+
Potato	--	+	+	--	--	+	-
Pepper	--	++	+/-	+/-	+/-	++	+
Cabbage	--	+	++	+/-		++	+
				country level			
Onion	--	+	+	+/-	--	+	+/-
Tomato	--	+	+	+	+/-	+	+
Potato	--	+/-	+/-	-	--	+/-	-
Pepper	--	+	++	+	+/-	+	+
Cabbage	--	+	+	+	-	+	+

Discussion

The results reflect the expected environmental impacts of freshwater use especially regarding different climates. Onion and pepper can be grown under dry and hot climates and hence need a lot of water in water-stressed areas. The relatively low impact from water use in tomato production arises from the relatively high impact from infrastructure expenditures, agrochemicals and partially heating, as it is mainly grown in greenhouses.

Regionalized assessment methods are crucial for agricultural production and should be further improved. We suggest using impact factors on watershed level rather than country averages, as this study shows that in larger countries national impact factors are not accurate. Not only regionalisation of inventory data and water use impact factors, but also regionalised impact assessment methods for other impact categories such as eutrophication and land use should be developed and applied in future studies in order to assess different production sites in a comprehensive way.

Single-score LCIA methods, such as EI99, are aggregating impacts of different categories based on subjective value judgements in the normalisation and weighting steps. However, these values might vary among different regions of the world, depending on the local culture and specific circumstances, especially for normalisation. EI99 allows differentiation of three cultural perspectives for impact assessment and weighting for coping with the problem of value judgements. Nevertheless, additional research on normalisation and weighting in the context of a regionalised, global methodology is necessary, as the EI99 method is developed for European conditions excluding regional aspects.

At current state of our research we are not able to systematically assess uncertainties in inventories or impact factors. Specification of uncertainty ranges is a crucial task for future, especially regarding additional uncertainties arising in spatially differentiated LCA studies. The combination of spatially explicit modelled foreground process with background process without spatial reference in LCA studies will be an additional methodological challenge.

Conclusion

The results of this work highlight the importance of integrating freshwater use in LCA studies on agricultural products and pinpoint the relevance of regionalisation on the level of the inventory analysis as well as impact assessment. Both aspects are crucial when comparing products with globalized value chains: for decision makers in food supply chains as well as for consumers interested in sustainable consumption.

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Proposing a life cycle land use impact calculation methodology

W.M.J. Achten[1], E. Mathijs[2], B. Muys[1]
[1] Katholieke Universiteit Leuven, Division Forest, Nature and Landscape, Celestijnenlaan 200 E Box 2411, BE-3001 Leuven, Belgium, wouter.achten@ees.kuleuven.be
[2] Katholieke Universiteit Leuven, Division Agricultural and Food Economics, Willem de Croylaan 42 Box 2424, BE- 3001 Leuven, Belgium

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Abstract

The Life Cycle Assessment (LCA) community is yet to come to a consensus on a methodology to incorporate land use in LCA, still struggling with what exactly should be assessed and which indicators should be used. To solve this problem we start from concepts and models describing how ecosystems function and sustain, in order to understand how land use affects them. Earlier our research group presented a methodology based on the ecosystem exergy concept. This concept as based on the hypothesis that ecosystems develop towards more effective degradation of exergy fluxes passing through the system and is derived from two axioms: the principles of (*i*) maximum exergy storage and the (*ii*) maximum exergy dissipation. This concept aiming at the area of protection natural environment is different from conventional exergy analysis in LCA focusing on natural resources. To prevent confusion, the ecosystem exergy concept is further referred to as the MAximum Storage and Dissipation concept (MASD concept). In this paper we present how this concept identifies end-point impacts, mid-point impacts and mid-point indicators. The identified end-point impacts to assess are Ecosystem Structural Quality (ESQ) and Ecosystem Functional Quality (EFQ). In order to quantify these end-point impacts a dynamic multi-indicator set is proposed for quantifying the mid-point impacts on soil fertility, biodiversity and biomass production (quantifying the ESQ) and soil structure, vegetation structure and on-site water balance (quantifying the EFQ). Further we present an impact calculation method suitable for different environmental assessment tools and demonstrate the incorporation of the methodology in LCA.

Introduction

Human activities have spatial needs for extraction of resources, forestry and agriculture, infrastructure and dwellings, industrial production processes and landfill. The use of land will often make the land unavailable for other uses, but may also change the quality of the land in terms of life support or potentiality for other land use (Heijungs *et al.* 1997; Lindeijer 2000; Lindeijer *et al.* 2002). In an LCA context land use was therefore defined (Lindeijer *et al.* 2002) as intensive human activities, aiming at exclusive use of land for certain purposes and adapting the properties of land areas in view of these purposes.

Land use and land use change are considered by the international community as a significant aspect of global change, which may induce climate change (Kalnay & Cai 2003; Lavy *et al.* 2004), desertification (Lavy *et al.* 2004; Asner & Heidebrecht 2005) and loss of biodiversity and life support functions (Lindeijer 2000; Lindeijer *et al.* 2002; Miles *et al.* 2004; Milà i Canals *et al.* 2007).

Several methods have been developed for the assessment of environmental impacts generated by land use and land use change (e.g. monitoring procedures, standards with principles, criteria and indicators (PC&I), environmental impact assessment (EIA) and life cycle assessment (LCA) (Baelemans & Muys 1998)). These methods and tools still face specific and shared problems regarding the land use impact assessment. Among these problems the selection and definition of relevant and measurable indicators seems one of the most persistent (Baelemans & Muys 1998). Discussions on land use impact in LCA community seem to reveal a lack of consensus on what exactly has to be assessed (Milà i Canals *et al.* 2006; Udo de Haes 2006; Baitz 2007; Milà i Canals *et al.* 2007; Milà i Canals 2007; Milà i Canals *et al.* 2007a). According to the authors the reason for these problems lies in the lack of a solid theoretical concept which can serve as paradigm in which land use and land use change impacts can be evaluated and assessed.

In this paper we propose a method to assess land use impact on the natural environment and life support functions (areas of protection). We propose to do this assessment from an ecosystem perspective, using a theoretical concept describing how ecosystems are structured and how they function. The rationale behind this starting point is, that we can only know how we damage an ecosystem by human induced land use if we understand how it works, lives and sustains. Based on the insight of this concept, we identify what exactly has to be assessed, translated in land use end-point impacts which should be assessed (also see (Peters *et al.* 2003; Garcia-Quijano *et al.* 2007b)). Based on published land use cause effect chains we propose a universally applicable (mid-point) indicator set. Since the links between the mid-point impacts and the end-point impacts are based on the theoretical concept the mid-point indicators are also compatible with the theoretical concept.

Background

Ecosystem theories can be divided in three groups: (*i*) succession models, (*ii*) resistance models and (*iii*) energy models. These latter combine the baseline of the succession models, which put most emphasis on internal control of the ecosystem, and the baseline of the resistance models, which put most emphasis on external control of the ecosystem. Energy models recognize the internal control of the self-organized complex system as a source of stability, but also considers the dependence of the ecosystem from external energy sources, which makes ecosystems stable only if they can sustain the bio-energetic control in case of external disturbances.

Among the energy models, the ecosystem exergy concept was introduced by Schneider & Kay (1994). According to them, ecosystems are open systems subject to continuous energy influxes. They tend to increase their internal exergy level, in order to evolve as far as possible from thermodynamic equilibrium. Doing so they develop towards more effective degradation of energy fluxes passing through the system. The concept is derived from two axioms: the principles of (*i*) maximum exergy storage and the (*ii*) maximum exergy dissipation (Fath *et al.* 2001). According to the maximum exergy storage principle an ecosystem on any site, with given abiotic features and local gene pool, would develop towards a state of highest possible exergy storage in terms of biomass, genetic information and complex structural networks (Jorgensen & Mejer 1979; Bendoricchio & Jorgensen 1997). The principal of maximum dissipation means that for any site an ecosystem would tend towards maximum dissipation of the exergy influxes in form of radiation, water, nutrients, air and genetics (Schneider & Kay 1995; Bendoricchio & Jorgensen 1997; Fath *et al.* 2001).The content of this ecosystem exergy concept is promising for further advances in land use impact. For a review on the ecosystem exergy concept see Dewulf *et al.* (2008).

It is important to stress that this concept, which aims at evaluating the area of protection of the natural ecosystem is different from conventional exergy analysis in LCA (Finnveden & Östlund P. 1997), which aims at accounting the use of natural resources. More on this topic can be found in Dewulf *et al.* (2008). In this paper we use the ecosystem exergy concept to justify the identification of the end-point, mid-point impacts and the indicator set used for quantification. To prevent from confusion with conventional exergy analysis, the authors will further refer to it as MAximum Storage and Dissipation concept (MASD concept), which stands for the succession and evolutionary trends observed in ecosystems (in modelling terms called goal functions), namely: (*i*) maximization of exergy storage in biomass, genetic information and structural networks (= maximization of Ecosystem Structural Quality, ESQ) and (*ii*) maximization of exergy dissipation from radiative, material and genetic influxes (= maximization of Ecosystem Functional Quality, EFQ, i.e. the buffering capacity which sustains the control of the ecosystem over the fluxes passing through it and its stability despite disturbances). These goal functions are interdependent of each other. Higher ESQ will lead to higher EFQ, which in turn will lead to further increase of the ESQ.

Approach

What should be assessed?

There is no agreement so far in the LCA community on what exactly should be assessed in the land use impact assessment. Based on the ecosystem concept explained above and the definition of land use (Lindeijer *et al.* 2002) we identify the end-point impacts which should at least be assessed.

In the light of the MASD concept the land use definition of Lindeijer *et al.* indicates that land use refers to human interventions bringing and keeping land at a certain Ecosystem Structural Quality (ESQ). In the MASD concept the affected ESQ will influence the Ecosystem Functional Quality (EFQ). Both goal functions are fundamental. Therefore we propose to assess the impacts on these two functions as being end-point impact of human land use interventions:

1. Impact on the Ecosystem Structural Quality (ESQ) (how does the human land use intervention influence the amount of living and dead biomass, the species composition and the complex ecosystem network structure?)
2. Impact on the Ecosystem Functional Quality (EFQ) (how does the human land use interventions influence the capacity of the land to keep control over solar energy, water, sediment and nutrients, to maintain and restore ESQ, and to buffer future disturbances?)

How to quantify the ESQ and EFQ indicators?

In order to quantify the ESQ and EFQ, relevant mid-point impacts of land use related interventions are selected, based on earlier published cause-effect chains (Köllner 2000; Lindeijer 2000; Lindeijer *et al.* 2002; Guinée *et al.* 2006) (the selection is given in Fig. 1). The list of mid-point impacts is non-exhaustive but, according to us, necessary to be assessed. Notice that we restrict ourselves to the land use interventions as human activities.

In a further step, the mid-point impacts have to be categorized to the end-point impacts (arrows in figure 1) and mid-point indicators have to be identified to quantify the mid-point impacts. This is an iterative process, since the content of the possible indicators determines the link between the mid-point and end-point (e.g. based on the explanation of the MASD concept, it might be expected that 'vegetation structure' should be categorized as ESQ, but the most suitable indicators quantifying the 'vegetation structure', namely leaf area index and vertical space distribution actually say more about the dissipation than about storage, see further). Furthermore, we aim (*i*) at proposing a simple impact score calculation method which is the same for each indicator (see further), (*ii*) at using easily available and/or measurable indicators and (*iii*) at selecting mid-point indicators representing four basic impact themes: soil, biodiversity, vegetation and water and that all themes contain indicators linked both to ESQ and EFQ.

Reference system land use change and land use occupation

The indicator values will give us a valuation of the ESQ and EFQ under a certain land use. An impact on ESQ and EFQ, caused by human induced land use change (LUCh), has to be measured against a reference system. The new installed land use ('Project LU'), should only be burdened for the change it makes compared to the land use it directly pushed away or will directly push away ('Former LU'), which, as such, should be the reference system (Fig. 2). For land use occupation (LUOcc) impact, the potential natural vegetation (PNV) is taken as reference. Since ESQ and EFQ are site specific, we propose to calculate the burdens (e.g. $ESQ_{Reference} – ESQ_{ProjectLU}$) relative (%) to the maximum potential ESQ and EFQ (or the PNV) of that specific location (Fig. 2). This reasoning will lead us further to an impact indicator calculation method (see further).

Following Lindeijer (Lindeijer 2000) the impact caused by land use change and by land use occupation is separated, because land use change can improve the land quality, compared with the situation before the change, but the land use occupation has still impacts on the maximization of storage and dissipation compared to absence of human induced land use. However, the land use occupation is seen as a quality difference between the maximal possible ESQ and EFQ (PNV) and the project ESQ and EBC.

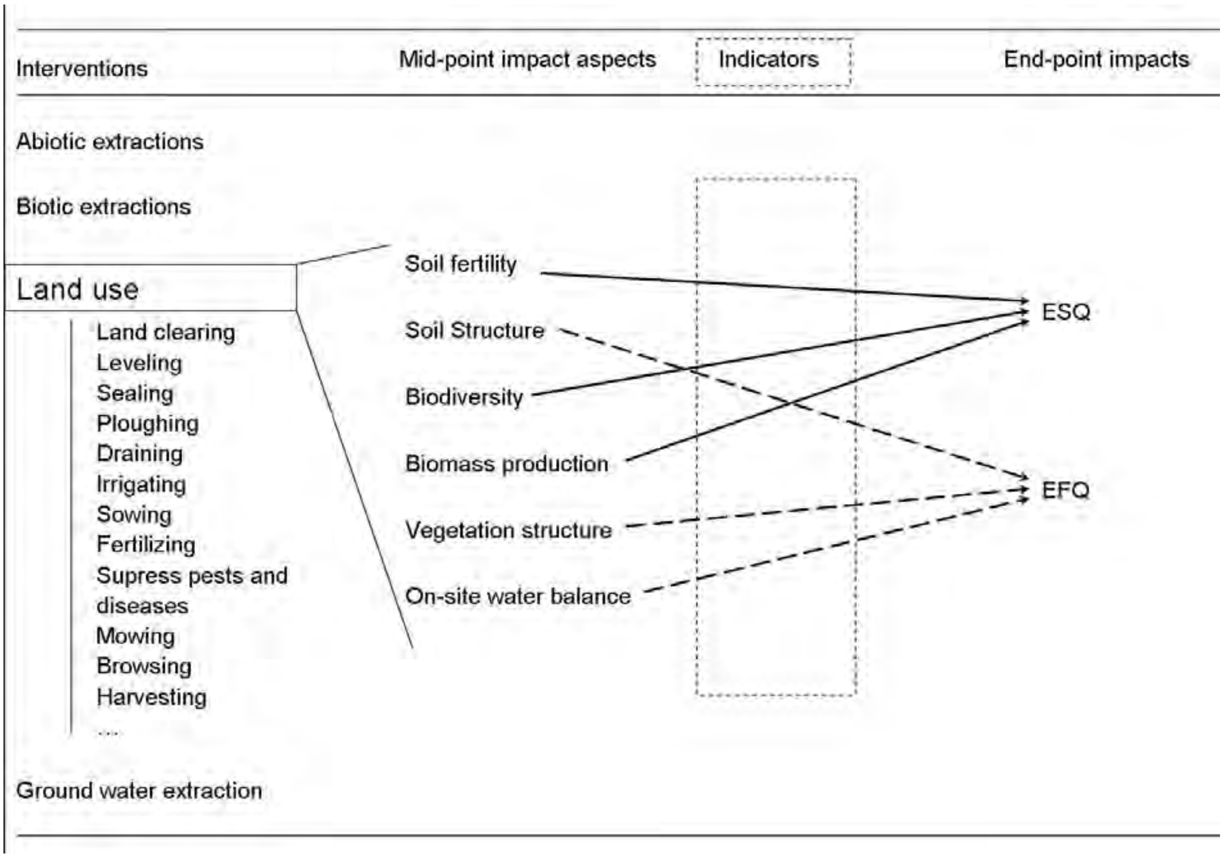

Fig. 1: Non-exhaustive overview of mid-point impacts of land use interventions. The arrows show the linkage of mid-point impacts with the end-point impacts.

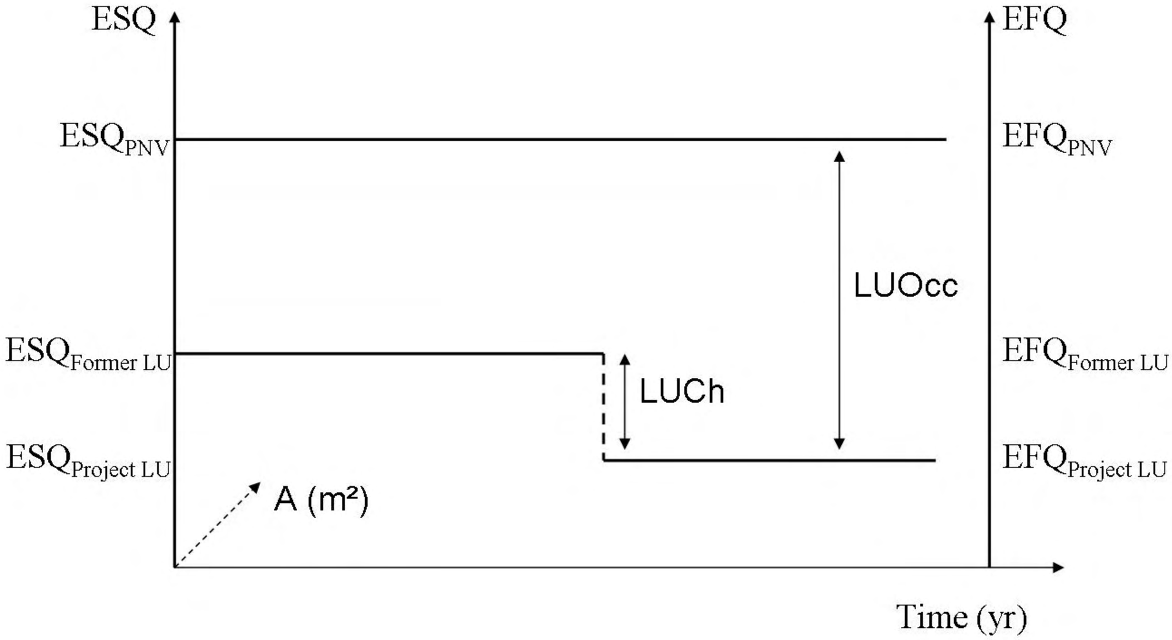

Fig. 2: Simplified depiction of land quality of the new induced land use (Project LU), former land use (Former LU) and potential natural vegetation (PNV).

Incorporation in LCA

The indicator set and the calculation method will give an environmental impact. From a LCA point of view these impacts should be reported per functional unit (FU) in order to be able to compare scenarios and managements around the world (Heijungs *et al.* 1997). Therefore we present a general formula for land use impact (S) calculation. This formula has two components: impact indicator component (I) and a LCA component (F) (Eq. 1).

$$S = I \times F \tag{Eq. 1}$$

Results

Impact indicator component

Set of mid point indicators

In this section a set of indicators is proposed. This set can be considered flexible. For each mid-point impact aspect two indicators are proposed, except for biodiversity. According to specific situations, specific aims of the user, data availability, measurement feasibility, etc. the users can choose to use both or just one. Further, there is still scope for extra possible indicators per mid-point aspect, according to users' expertise.

Indicators quantifying ESQ

Soil fertility

For assessing impact on soil fertility two indicators are proposed: (*i*) cation exchange capacity (CEC) and (*ii*) base saturation (BS) of the topsoil (0-30 cm). CEC has a direct impact on the soil ability to support vegetation and therefore on the ability of the ecosystem to produce and store biomass (Esthetu *et al.* 2004; Rutigliano *et al.* 2004; Bronick & Lal 2005). Loss of BS is considered an impact because it decreases the ecosystem productive capacity and therefore its capacity to store biomass and genetic information (Hagen-Thorn *et al.* 2004). Both CEC and BS are directly affected negatively or positively by management practices (Johnson 2002; Favre *et al.* 2002; Lyan & Gross 2005; Asano & Uchida 2005). Both CEC and BS require on-field measurements with standard chemical analysis of soil samples.

Biomass production

Any decrease of biomass due to harvest in any of its forms or by changes in site quality is assumed to cause a decrease of ecosystem control over energy (e.g. radiation), nutrients and water flows (Mortimore *et al.* 1999; Houghton & Hackler 1999; Son *et al.* 2004; Scheller & Mladenoff 2005; Kettunen *et al.* 2005). Therefore the proposed indicators look at the (*i*) total above biomass (TAB) and (*ii*) free net primary production (fNPP). Net primary production (NPP) is controlled by physical, environmental and biotic factors (Garcia-Quinjano & Barros 2005). fNPP is the part of NPP which is not harvested but stays in the ecosystem to fulfil life support functions (Lindeijer 2000). fNPP data is available on a world-wide scale (Lindeijer 2000), TAB is best measured on the field.

Species diversity

Based on the same reasoning of data availability as Lindeijer (Lindeijer 2000) we opted for vascular plant species number as sole biodiversity indicator. This indicator required on-field measurements.

Indicators quantifying EFQ

Soil structure

Impacts on soil structure can be assessed by: (*i*) soil organic matter (SOM) of the topsoil (0-30 cm) and (*ii*) soil compaction. SOM is an good indicator of the dynamic nature of soils (Mila i Canals *et al.* 2007b) and for the physical and chemical filter and buffer capacity (Milà i Canals 2003). Soil compaction reduces the volume of air in the soil and reduce infiltration rate and as such can have negative impacts on root development and biomass production (Munkholm *et al.* 2005) and increased surface runoff (Jonson-Maynard *et al.* 2002; Green *et al.* 2003). In Fig. 1 the soil structure impact aspect is characterized as impact on EBC, Therefore infiltration rate is used as soil compaction indicator (I) (see further). This indicator will highlight changes in the capacity of the ecosystem to buffer water and sediment flows. SOM is easily available (Mila i Canals *et al.* 2007b) while I is best measured in the field.

Vegetation structure

Characterized to EBC, the proposed indicators are (*i*) leaf area index (LAI) and (*ii*) vertical space distribution. LAI is a reliable indicator of a systems absorption capacity of solar radiation (Rascher *et al.* 2004; Dungan *et al.* 2004), systems reduction potential of kinetic energy from raindrops (Anzhi *et al.* 2005)(Van Dijk & Bruijnzeel 2001; Gomez *et al.* 2001; Pañuelas *et al.* 2003) and systems interception and retention of rainwater (Schellekens *et al.* 1999; Cuartas *et al.* 2007; Komatsu *et al.* 2007). Vertical space distribution, calculated by dividing the canopy height of the dominant stratum of the land use (H) by the number of vertical strata in the land use (S), gives an idea about the vertical structure of the vegetation interface buffering solar radiation, rainfall, wind, among others flows. For the same height of the dominant layer in the vertical structure, a lower number of layers would decrease the optimal or maximum buffer capacity of the ecosystem (Onaindia *et al.* 2004; Will *et al.* 2005; Wehrli *et al.* 2005; Stephens & Gill 2005). A LAI global 1 km geodataset is available at the Land Processes Distributed Active Archive Centre (LP DAAC, USA) (https://lpdaac.usgs.gov/), but can also be measured in the field by hemispheric photography. Vertical space distribution is best measured in the field.

On-site water balance

Here evapotranspiration and soil cover are proposed. Loss of evapotranspiration level indicates a decrease of health and productivity of the ecosystem and a loss of control over energy, water and material flows (Obrist *et al.* 2003; Goyal 2004). Note that this is only used as on-site indicator. Off-site effects (on aquatic systems) of changing ET are not considered (see discussion). Soil cover (0-30 cm above ground level) is seen as an indicator of buffer capacity for raindrop impact and superficial erosion (Morgan 1995). Data on both of these indicators are available in geodatasets of LP DAAC, USA. Soil cover is also measurable on-field.

Impact calculation

The impact indicator scores (IS) are the summation of the relative impacts of the different land use activities of which a certain project or production process consists multiplied by the relative area of the activity (A_i) (i.e. area of the activity under evaluation over the total area use of the project (A_t)). The relative impacts are the difference between the observed indicator value and the indicator value for the reference system (for the impact calculation of the land use change the reference system is the former land use, for impact of the land use occupation the reference system is the PNV), normalized by the indicator value of the potential natural vegetation (PNV) in the region. To express the product in percentage it is multiplied by 100 (Eq. 2).

$$IS = \sum_i \left(\frac{A_i}{A_t} * \frac{\left[Value_{ref} - Value_{proj,i} \right]}{Value_{PVN}} \right) * 100 \qquad \text{Eq. 2}$$

with A_i is the area of the specific activity under evaluation, A_t is the total area of the project site, $Value_{proj,i}$ is the value for the selected indicator for the project area of the specific activity under evaluation and $Value_{ref}$ is the value of the selected indicator for the reference system (i.e. former land use for land use change and PNV for land use occupation).

Table 1 gives an overview of the proposed indicators per mid-point impact aspect and the corresponding score calculation for land use change and land use occupation. Indicators and formula are chosen in such way that negative environmental impacts give a positive indicator score.

Based on these impact indicator calculations the impact indicator component for structural and functional land quality change due to land use occupation can be calculated.

$$I_{ESQ} = \frac{\overline{IS_{Sf}} + \overline{IS_{\alpha-Bd}} + \overline{IS_{Bp}}}{3} \qquad \text{Eq. 3}$$

$$I_{EFQ} = \frac{\overline{IS_{Ss}} + \overline{IS_{Vs}} + \overline{IS_{Wb}}}{3} \qquad \text{Eq. 4}$$

with I the impact indicator component and $\overline{IS_x}$ the average indicator score for mid-point impact aspect *x* (Sf = Soil fertility; α-Bd = On site biodiversity; Bp = Biomass production; Ss = Soil structure; Vs = Vegetation structure and Wb = On site water balance) (Tab. 1). Eq. 3 and 4 will result in relative impacts on the land system structure and land system functioning expressed in percentages.

LCA component

The LCA component (F) is necessary to present the impacts per FU. We propose to use the following F (Eq. 5) for both LUCh and LUOcc.

$$F = \frac{(time * area)}{FU} \qquad \text{Eq. 5}$$

Where *FU* is the functional unit of the project or production process and (*time**area*)* is the area needed to produce a *FU* for a specific period of time.

Discussion

This paper mainly aims to provide another approach to solve some general problems in land use impact assessment. Starting from a concept (MASD) which explains how, through ecosystem functions, an ecosystem works, lives and survives, we identified meaningful end-point impacts of human land use impacts. In the light of the MASD concept cause effect chains and possible mid-point indicators from literature were interpreted, leading to a balanced selection of a set of easily available or measurable mid-point indicators. Our proposal contains a dynamic use of our indicator set, where the user can argument to use only a minimum set of six indicators or to add specific indicators. The fact that for each mid-point impact, except soil fertility, data is available for at least one indicator, strengthens the dynamic and workable nature of this indicator set. The fact that averages of the mid point indicators are used downstream the calculation, overlap between the two selected indicators is not a problem Furthermore this indicator set gives a balanced look on basic impact themes: soil, water, vegetation and biodiversity.

Starting the approach from a general founding paradigm makes the proposed end-point impacts and indicator set applicable in different kinds of assessment tools, including LCA, as described in this paper (see LCA component).

The calculation of the land use change and occupation impact between the respective reference land use and the project land use relative to the local PNV results in a non site-specific impact (%). As the impact is actually scaled against the maximum possible, the impact does not contain impacts of land use changes or occupations prior to the land use of interest of the LCA study.

Although this proposal contains improvements of earlier work (Peters *et al.* 2003; Garcia-Quijano *et al.* 2007a) there is still scope for improvement. (*i*) Currently off-site impacts are not considered. There is a clear need for addressing off-site effects on biodiversity and water balance (but see (Heuvelmans *et al.* 2005)). (*ii*) The aggregation of the mid-point impacts into the end-point impacts is done using equal weighting. This is because of lack of information on the respective importance of the different variables in the ecosystem goal functions.

Tab. 1: Proposed indicators per mid-point impact aspect and impact score calculation for land use change and land use occupation

Mid-point	Indicator(s)		LUCh	LUOcc
Soil fertility	Cation exchange capacity (CEC)	IS_{Sf}	$\sum_i \left(\frac{A_i}{A_t} * \frac{(CEC_{ref} - CEC_{proj,i})}{CEC_{PNV}} \right) * 100$	$\sum_i \left(\frac{A_i}{A_t} * \frac{(CEC_{PNV} - CEC_{proj,i})}{CEC_{PNV}} \right) * 100$
	Base saturation (BS)	IS_{Sf}	$\sum_i \left(\frac{A_i}{A_t} * \frac{(BS_{ref} - BS_{proj,i})}{BS_{PNV}} \right) * 100$	$\sum_i \left(\frac{A_i}{A_t} * \frac{(BS_{PNV} - BS_{proj,i})}{BS_{PNV}} \right) * 100$
Soil structure	Soil organic matter (SOM)	IS_{Ss}	$\sum_i \left(\frac{A_i}{A_t} * \frac{(SOM_{ref} - SOM_{proj,i})}{SOM_{PNV}} \right) * 100$	$\sum_i \left(\frac{A_i}{A_t} * \frac{(SOM_{PNV} - SOM_{proj,i})}{SOM_{PNV}} \right) * 100$
	Soil compaction (Infiltration rate, I)	IS_{Ss}	$\sum_i \left(\frac{A_i}{A_t} * \frac{(I_{ref} - I_{proj,i})}{I_{PNV}} \right) * 100$	$\sum_i \left(\frac{A_i}{A_t} * \frac{(I_{PNV} - I_{proj,i})}{I_{PNV}} \right) * 100$
Biomass production	Free net primary production (fNPP)	IS_{Bp}	$\sum_i \left(\frac{A_i}{A_t} * \frac{(fNPP_{ref} - fNPP_{proj,i})}{fNPP_{PNV}} \right) * 100$	$\sum_i \left(\frac{A_i}{A_t} * \frac{(fNPP_{PNV} - fNPP_{proj,i})}{fNPP_{PNV}} \right) * 100$
	Total aboveground biomass (TAB)	IS_{Bp}	$\sum_i \left(\frac{A_i}{A_t} * \frac{(TAB_{ref} - TAB_{proj,i})}{TAB_{PNV}} \right) * 100$	$\sum_i \left(\frac{A_i}{A_t} * \frac{(TAB_{PNV} - TAB_{proj,i})}{TAB_{PNV}} \right) * 100$
Vegetation structure	Leaf area index (LAI)	IS_{Vs}	$\sum_i \left(\frac{A_i}{A_t} * \frac{(LAI_{ref} - LAI_{proj,i})}{LAI_{PNV}} \right) * 100$	$\sum_i \left(\frac{A_i}{A_t} * \frac{(LAI_{PNV} - LAI_{proj,i})}{LAI_{PNV}} \right) * 100$
	Vertical space distribution (ratio of canopy height of the dominant strata (H) devided by number of strata (St))	IS_{Vs}	$\sum_i \left(\frac{A_i}{A_t} * \frac{\left(\frac{H_{ref}}{St_{ref}} - \frac{H_{proj,i}}{St_{proj,i}} \right)}{\frac{H_{PNV}}{St_{PNV}}} \right) * 100$	$\sum_i \left(\frac{A_i}{A_t} * \frac{\left(\frac{H_{PNV}}{St_{PNV}} - \frac{H_{proj,i}}{St_{proj,i}} \right)}{\frac{H_{PNV}}{St_{PNV}}} \right) * 100$
On-site water balance	Evapotranspiration (ET)	IS_{Wb}	$\sum_i \left(\frac{A_i}{A_t} * \frac{(ET_{ref} - ET_{proj,i})}{ET_{PNV}} \right) * 100$	$\sum_i \left(\frac{A_i}{A_t} * \frac{(ET_{PNV} - ET_{proj,i})}{ET_{PNV}} \right) * 100$
	Soil cover (SC)	IS_{Wb}	$\sum_i \left(\frac{A_i}{A_t} * \frac{(SC_{ref} - SC_{proj,i})}{SC_{PNV}} \right) * 100$	$\sum_i \left(\frac{A_i}{A_t} * \frac{(SC_{PNV} - SC_{proj,i})}{SC_{PNV}} \right) * 100$
Biodiversit y (on site α diversity)	Species diversity (Number of vascular plant species (NS))	$IS_{Bd-\alpha}$	$\sum_i \left(\frac{A_i}{A_t} * \frac{(NS_{ref} - NS_{proj,i})}{NS_{PNV}} \right) * 100$	$\sum_i \left(\frac{A_i}{A_t} * \frac{(NS_{PNV} - NS_{proj,i})}{NS_{PNV}} \right) * 100$

In addition to the link with the FU (LCA component), there is scope to include a temporal dimension in Eq. 1. This is particularly interesting in case of an impact fluctuating over time and consists of integrating the impact over time. This implies knowledge of how an impacting factor would intervene in the long term dynamics of an ecosystem. Therefore, calculation of this component will depend on the state of knowledge and on data availability.

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A new LCIA method for assessing impacts of agricultural activities on biodiversity (SALCA-Biodiversity)

Ph. Jeanneret[1], D.U. Baumgartner[1], R. Freiermuth Knuchel[1], G. Gaillard[1]
[1]Agroscope Reckenholz-Tänikon Research Station ART, CH-8046 Zurich, Switzerland,
philippe.jeanneret@art.admin.ch

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Abstract

Agroscope Reckenholz-Tänikon Research Station ART developed a method for the integration of biodiversity (organismal diversity) as an impact category of Life Cycle Assessment (LCA) for agricultural production (SALCA-Biodiversity). This method is valid for grasslands and arable crops, and integrates semi-natural habitats of the farming landscape to estimate the impact of management systems on biodiversity. First, a list of 11 indicator species groups (flora, birds, mammals, amphibians, snails, spiders, carabids, butterflies, wild bees, and grasshoppers) was established considering ecological and life cycle assessment criteria. Second, inventory data about agricultural practices with detailed management options were specified. Third, a scoring system estimated the reaction of every indicator species group regarding management options, followed by aggregation steps. In a case study, biodiversity scores for grassland along an intensity gradient as well as winter wheat with differing cropping systems were calculated. Results showed the dominant influence of management and production intensity on most indicators and management options from which large impacts on biodiversity are to be expected. The use of 11 indicator species groups allows a differential and a fairly comprehensive estimation of the impacts of the agricultural practices on biodiversity. With SALCA-Biodiversity, production systems can be compared regarding their potential impact on biodiversity, and may therefore help in making recommendations for good practices.

Introduction

Currently, the necessary integration of biodiversity and/or land use as impact category in Life Cycle Assessment (LCA) methodologies is recognized (SETAC/UNEP LCA Initiative, Milà i Canals *et al.* 2007). In this context, Agroscope Reckenholz-Tänikon Research Station ART developed a method for the integration of biodiversity as an impact category for Life Cycle Assessment (LCA) of agricultural activities (SALCA-Biodiversity, Jeanneret *et al.* 2006). Two approaches for evaluating the effects of agricultural activities (in a broad sense) on biodiversity are found in the literature: (1) biodiversity is included as a mid-point impact category in LCA like other categories, e.g. the global warming potential. This approach is essentially based on the species diversity of vascular plants and includes the impact of industry, agriculture and transport on a continent scale (e.g. Lindeijer *et al.* 1998, Müller-Wenk 1998, Köllner 2000, Milà i Canals *et al.* 2007) and also evaluates the rarity of the ecosystems and their vulnerability (Weidema & Lindeijer 2001). (2) An environmental diagnosis based on a biotope evaluation with indicators is performed ("ecological value" of farms, e.g. Frieben 1998, Brosson 1999).

Our method is based on the first approach with two characteristics distinguishing it from methods published so far:

- A detailed focus on agricultural activities. The method is designed to be used in combination with conventional mid-point LCIA methods (see for example Nemecek *et al*. 2005). Since the impact on biodiversity is area specific, the use of SALCA biodiversity in comparative LCA requires that the same occupation in terms of area and duration is satisfied by both systems compared. In other words, in case where a product related functional unit is used and the systems compared have different area yields, it is necessary to complement the analysed systems in such a way that the same area during the same period is cultivated.

– A thorough consideration of species groups affected in their diversity (i.e. flora and fauna), the present parameterisation being valid for use in Switzerland and adjoining regions. Of course, complex biodiversity in the broadest sense of the Rio Convention cannot be totally measured as such. However, a single indicator is unlikely to be devised even in agro-ecosystems that surrogate for all other organisms with respect to reaction to farming operations (e.g. Büchs 2003). Instead, groups of indicators shall be selected that are sensitive to environmental conditions resulting from land use and farming operations, and give as representative a picture as possible of biodiversity as a whole.

The method presented aims at estimating and comparing the impact of agricultural management systems on biodiversity by using a set of indicator species groups. In a specific case study, results of the application of the method to several scenarios representing field management options for grassland (intensity level) and wheat (cropping system) were calculated for illustration.

Materials and methods

In the present method the choice of indicator species groups (ISGs) was made using a criteria table based on the linking of the species to agricultural activity, and general criteria such as the species distribution in the cultivated landscapes, their habitats and their place in the food chain (Jeanneret *et al.* 2006). Although recognized as a very important habitat for biodiversity supporting a high number of functions, soil and soil organisms have not been considered in this method. The reason is that impacts of agricultural practices on biodiversity in soil have not been sufficiently investigated. Then, the following ISGs were selected: flowering plants (grassland and crop flora), birds, small mammals, amphibians, snails, spiders, carabid beetles, butterflies, wild bees and grasshoppers. Furthermore, we distinguished between the overall species diversity of each species group and the ecologically demanding species (stenotopic species, red list species) in the impact estimation.

The detailed effects of the management activities on each ISG were estimated based on information from the literature and expert knowledge. Most of the impact of specific management activities on indicator species groups are known and published in scientific papers. For example, the impact of the number of cuts of a meadow on butterfly species (e.g., Erhardt & Thomas 1989, Feber *et al.* 1996, Gerstmeier & Lang 1996); the impact of cultivation practices on carabid beetles(Clark *et al.* 1997, Hance 2002, Holland 2002) are described. This information was discussed and completed by experts before entering the rating system (Tab. 1). In this study, all the typical management activities of grassland and winter wheat fields such as manuring, mowing, insecticide and fungicide applications were specified with options, e.g. the type of fertiliser and the mowing period, the type of insecticide and fungicide and the application period (restricted to the Swiss farming). The impact of each management option on ISGs was rated on a scale of 0 to 5 (rating *R*, Tab. 1).

Tab. 1: Rating *R* of management option impact on the selected indicator species groups (ISG).

0:	The species group is unaffected because it does not occur in the considered agricultural habitat.
1:	The option leads to a severe impoverishment of species diversity within the species group considered and renders impossible the occurrence of stenotopic species and red list species.
2:	The option leads to a slight impoverishment of species diversity within the species group considered and renders impossible the occurrence of stenotopic species and red list species.
3:	The option has no direct effect on the species group considered.
4:	The option leads to a slight increase in species diversity within the species group considered and makes possible the occurrence of stenotopic species and red list species.
5:	The option promotes species diversity within the species group considered and makes possible the occurrence of stenotopic species and red list species.

Since agricultural habitats of the farming landscape have not the same suitability with respect to specific ISG, a coefficient ranging from 1 to 10 ($C_{habitat}$) was attributed to weight the rating of the management options for each ISG specifically. Similarly, a second coefficient from 0 to 10 ($C_{management}$) quantified the relative importance of management activities for a given habitat, e.g. grazing and mowing in grasslands, manuring and pesticide application in winter wheat, for each ISG. The final score *S* of a management option was the product of the rating of the management option *R* and the mean value of the two weighting coefficients $C_{habitat}$ and $C_{management}$ ($S = R * C_f$; where S = final rating, R = impact rating of a management option and C_f = final coefficient = $[C_{management} + C_{habitat}]/ 2$). In case of management activities repeated during the year (e.g. mowing) an annual average was calculated when the ISG can recover from one period to another, or the most negative period was considered in case of a permanent damage. The final ISG score of a given agricultural habitat was calculated as the mean *S* over the management options. Furthermore, ISG scores were aggregated to a biodiversity score by weighting each ISG score on the basis of trophic links between the ISGs and species richness of the ISG. The more important an ISG as a basic food for other indicators and the more species-rich in the cultivated landscapes of Switzerland, the higher its weighting. Comparison of management scenarios can then be made at field level first but as ratings and coefficients were also defined for semi-natural habitats, ISG and biodiversity scores can also be calculated at farm level by aggregation of the scores obtained for single agricultural habitats (except vegetable, fruit and grape crops).

To illustrate use of the method and discuss results of impact calculation on biodiversity and particular ISGs, realistic scenarios of grassland and winter wheat management systems for the Swiss lowlands were defined (Tab. 2, Nemecek *et al.*, 2005). Scenarios addressed a large intensity gradient for grasslands ranging from one utilization and no fertilization (2.7t DM/ha and year) to five utilizations and fertilizer applications (11t DM/ha and year). Similarly, various cropping systems were considered for winter wheat along a gradient of production intensity (3.5t DM/ha and year – 5.8t DM /ha and year).

Tab. 2: Management characteristics and production of grassland and winter wheat systems used to test the method of impact calculation on ISGs.

Grassland systems (hay production)		Management characteristics and production
A	Intensive grassland	5 cuts/year, fertilised with slurry; 11t DM/ha
B	Fairly intensive grassland	4 cuts/year, fertilised with slurry; 9t DM/ha
C	Low intensive grassland	3 cuts/year, fertilised with solid manure; 5.6t DM/ha
D	Extensive grassland	1 cut/year; no fertilisation; 2.7t DM/ha
Winter wheat systems		
E	Conventional production	5.8t DM/ha
F	Integrated production– intensive	5.5t DM/ha
G	Integrated production – extensive	4.5t DM/ha
H	Organic production	3.5t DM/ha

Results

Compared results of grassland and winter wheat systems suggested that the crop was on average less suitable for most of the ISGs (Tab. 3). The transition from conventional and intensive integrated winter wheat systems (scenario E and F) to extensive (integrated) and organic production (scenario G and H) did not reveal the spectacular increase of scores occurring from intensive and fairly intensive (A and B) to low intensive and extensive grassland systems (C and D). However, conventional and integrated winter wheat systems (E and F) exhibited slightly higher aggregated biodiversity scores than the most intensive managed grasslands (A and B). This difference was mainly due to higher scores obtained by the crop flora (compared to the grassland flora) and the carabid beetles as shown by detailed ISG results. The highest scores were calculated for butterflies in extensive grassland and the crop flora in winter wheat, 36.0 (D) and 17.3 (H), respectively, and the lowest for amphibians in intensively managed grassland and winter wheat, 0.8 (A and B) and 1.4 (F), respectively. For a rough comparison, the aggregated biodiversity score obtained by a hedgerow with a standard management (result not shown), as a typical semi-natural habitat of the agricultural landscape, is about 21, and varies between 11 and 38 depending on ISG.

Calculated for the range of grassland types, scores definitely increased with decreasing management intensity (scenarios A to D) for the aggregated biodiversity, the overall species diversity of most of the ISGs and for the ecologically demanding species (Tab. 3). Scores for ecologically demanding species were slightly lower than those of overall species diversity. An obvious inflection point occurred between 4 and 3 cuts/year (fairly intensive and low intensive grasslands) and a change of the manure form. Indeed, aggregated biodiversity scores increased by 0.2 from intensive to fairly intensive, by 7.4 from fairly intensive to low intensive. Nevertheless, scores increased by an additional 7.5 from low intensive to extensive grasslands. Snails were an exception to this pattern, the largest difference taking place between low intensive and extensive grassland (93.9% increase). No fertilization at all was then more important than the fertilizer form for snails. Extensive grasslands obtained higher biodiversity scores than low intensive grasslands except for mammals which do not take advantage of one of both types. The largest difference in percentage occurred between fairly intensive and low intensive grasslands for the amphibian special life phase but at a very low score level (aquatic life phase, 0.8 to 2.9, 262.5%). The highest scores were obtained by butterflies in extensive grasslands (36.0 for the overall diversity and the ecologically demanding species), followed by grasshoppers and wild bees.

Regarding winter wheat systems, organic production obtained the highest aggregated biodiversity and ISG scores. Aggregated biodiversity scores increased stepwise slowly, from the intensive integrated production (reference scenario), to the organic production, i.e. F to E, 0.2 (2.7%), E to G, 0.7 (9.1%), G to H, 0.3 (3.6%). Interestingly, spiders and birds showed the highest increase of scores from conventional (E) to extensive integrated production (G) with 2.3 (28%) and 0.9 (17%), respectively, and 2.3 (28.8%) for ecologically demanding spider species. The lowest scores were calculated for amphibians, snails and mammals, for which change of production system only causes minor changes of scores. Conventional production obtained a slightly higher score for wild bees at a relatively low level (5.2), however. For grassland flora, butterflies and grasshoppers, no scores were calculated because crop fields have no or negligible importance as habitat for these ISGs.

Tab. 3: Results of SALCA-Biodiversity for grassland and winter wheat systems. ISG and biodiversity scores are given per ha cultivated crop. Scores of grassland system (A) and winter wheat system (F) are set as reference scores. Scores with the same format are considered similar to the reference (95%< score <104%). Scores underlined are considered better than the reference (105%< score <114%). Scores double underlined and bold are considered much better than the reference (score >115%). Theoretical minimum score is 1 and maximum 50. No scores means no relevance for the considered system.

	Biodiversity scores							
	Grassland				Winter Wheat			
Production systems	A	B	C	D	E	F	G	H
	Overall species diversity							
Aggregated[1]	6.2	6.4	**13.8**	**21.3**	7.7	7.5	8.4	**8.7**
Grassland flora	3.7	3.9	**11.4**	**18.5**				
Crop flora					15.2	15.1	16.0	17.3
Birds	6.4	6.7	**13.8**	**22.0**	5.3	5.0	**6.2**	**6.4**
Mammals	7.3	7.3	**11.1**	**11.1**	4.6	4.6	4.6	4.6
Amphibians	2.1	2.1	**5.2**	**9.5**	1.7	1.7	1.8	1.8
Snails	5.4	5.6	5.8	**11.3**	2.2	2.2	2.2	2.2
Spiders	9.1	9.3	**15.8**	**22.4**	8.2	8.0	**10.5**	**10.7**
Carabid Beetles	7.0	7.4	**13.6**	**21.0**	10.9	10.6	11.7	11.9
Butterflies	6.8	7.0	**20.0**	**36.0**				
Wild Bees	7.4	7.6	**18.6**	**23.0**	5.2	4.9	5.0	4.8

	Biodiversity scores							
	Grassland				Winter Wheat			
Production systems	A	B	C	D	E	F	G	H
Grasshoppers	6.9	6.9	**19.4**	**33.1**				
	Ecologically demanding species							
Amphibians	0.8	0.8	**2.9**	**4.8**	1.5	1.4	1.6	1.6
Spiders	8.9	9.0	**15.3**	**21.6**	8.0	7.8	**10.3**	**10.5**
Carabid Beetles	7.0	7.3	**13.4**	**20.6**	10.6	10.1	11.2	11.3
Butterflies	6.7	6.8	**19.4**	**36.0**				
Grasshoppers	6.8	6.8	**19.3**	**32.9**				

[1]ISG scores are aggregated taking into account rules of trophic relations between indicator species groups.

Discussion

Aggregated biodiversity and ISG scores suggest that biodiversity is on average less impacted by grassland than by winter wheat systems. This can be explained by a higher wide-ranging disturbance level usually occurring in crop fields compared to grasslands. However, the difference between grassland and winter wheat mainly occurred in less productive systems, i.e. in extensive and low intensive grassland compared to extensive integrated or organic production of winter wheat. The reason is that a crop field remains a monoculture with low habitat diversity even in extensively managed systems. In the contrary, grasslands with extensive management usually encompass large habitat diversity by first providing species-rich vegetation. The spectacular scores obtained by most of the IGSs in the extensive grassland system showed the importance of this management for biodiversity. The scores distinctly decreased in two steps, first from extensive to low intensive grassland, and then from low intensive to fairly intensive and intensive grassland, demonstrating that impacts occurred due to the increasing number of cuts (3 to 4-5 cuts/year and 1 to 3 cuts/year), which directly affects the habitat, and the fertilisation form. The high scores for butterflies, grasshoppers and wild bees in extensively used grassland were mainly due to the high habitat coefficients attributed to grassland habitats reflecting their importance for all three ISGs in the agricultural landscape as potential habitat. Detailed analysis of results also showed that dramatic effect can be observed by increasing the management intensity and increasing the production level accordingly, from low intensive to fairly intensive grasslands (115.6% decrease of the aggregated biodiversity score).

Although at a lower level than extensively managed grassland, organic production obtained the highest scores for the aggregated biodiversity and ISG scores among winter wheat systems. This is in accordance with the management techniques that usually take place in this system, and their impact on ISG, i.e. no application of chemical-synthetic pesticides and lower fertilization rate. Compared to its extensive form, the intensive integrated production negatively affected in particular spiders and birds because of the use of unselective pesticides and the more frequent disturbances involved for usual farming operations.

Conclusion

Although limited to agriculture, the method SALCA-Biodiversity represents an important step toward integration of biodiversity in LCA. With SALCA-Biodiversity, impacts of the most important agricultural practices and choices of farmers on biodiversity can be recognized. Impacts of agricultural practices on several indicator species groups of the above-ground habitats that take place in grassland and crop systems can be compared and recommendations can be made accordingly. Results showed that impacts are specific to indicator species groups and cannot reliably be derived from one single indicator.

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Ecoinvent-based extrapolation of crop life cycle inventories to new geographical areas

T. Nemecek & T. Kägi
Agroscope Reckenholz Tänikon Research Station ART, Reckenholzstrasse 191, CH-8046 Zürich, phone +41 44 377 72 54, fax +41 44 377 72 01, thomas.Nemecek@art.admin.ch

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Abstract

LCA practitioners are often confronted with the situation where life cycle inventories are available only for other regions than the one under study. The question arises if and how LCI data can be extrapolated from one region to another. The database ecoinvent V2.01 contains several inventories for the same crop in different countries showing considerable differences. Life cycle inventories and environmental impacts depend on pedo-climatic conditions, crop management and yields. All three factors are shown to be highly variable, not only between countries but also within a country. The analysis of the wheat example shows that the yield is a determining factor for all environmental impacts and shows a close correlation to the land occupation and to environmental impact assessment methods highly dependent on it like EcoIndicator '99. Fertilisation is shown to be a key factor for many environmental impacts, particularly global warming, eutrophication and acidification. A high correlation was found between the amount of nitrogen fertiliser and CED. Pesticide applications strongly influence ecotoxicity and human toxicity, while the use of the machinery has a moderate impact only on energy demand and ozone formation. Extrapolation from existing datasets was possible to some extent for EcoIndicator '99 and for cumulated energy demand.

Introduction

Compared to the version 1.3, the version 2.0 of the ecoinvent database (eiV2.0) contains new inventories of agricultural products, in particular for a given crop in various production places or countries (Nemecek & Kägi 2007) and inventories of biomass production for energy uses (Jungbluth *et al*. 2007). A comparison of the environmental impacts between different production places reveals considerable differences (Fig. 1). In other words: by simply using an inventory under conditions other than those under which it was defined, the impacts under- or overestimated by a factor of four.

The question arises, whether it is possible to extrapolate life cycle inventory and impact data from one region to another.

First, we will look at the relationships influencing the environmental impacts of crop production and how the different influencing factors vary in the crop production regions. In the second step, we will analyse the key factors for the environmental impacts. In the last section, we will analyse the wheat datasets in ecoinvent V2.01 to see by example, how datasets could be extrapolated to other geographical regions.

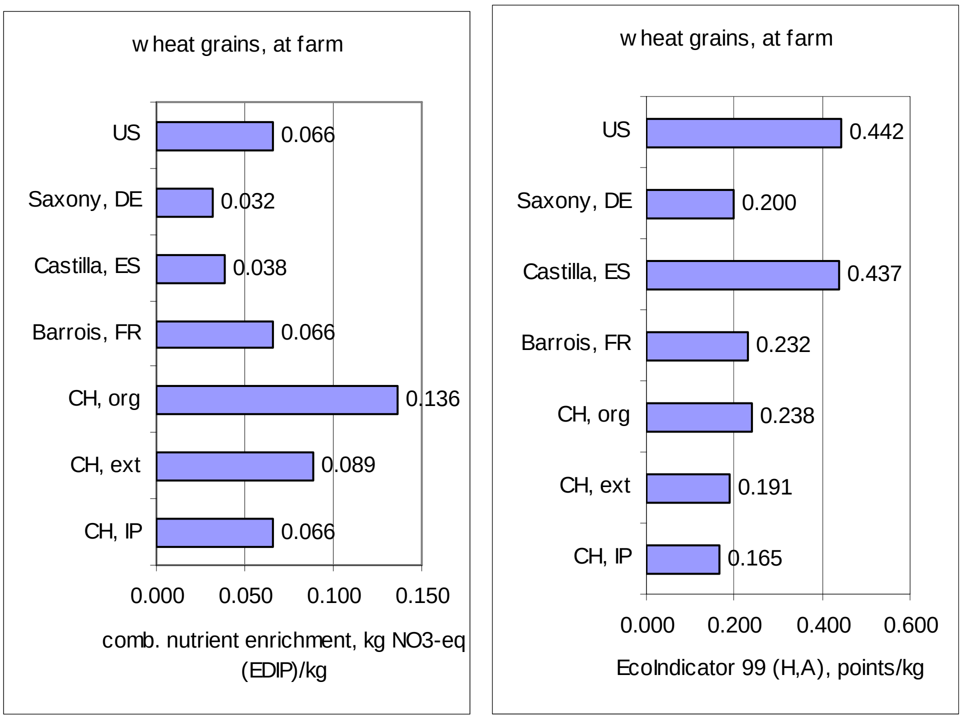

Fig. 1: Combined nutrient enrichment potential according to EDIP97 and EcoIndicator99 (HA) points for different wheat production inventories (ecoinvent data V2.0, ecoinvent Centre, 2007).

Relationships influencing the environmental impacts and variability of production conditions

First, we need to understand how the environmental impacts of a crop product are influenced. The pedo-climatic conditions (i.e. the natural environment) influence both the crop management and the yield (Fig. 2). For example, irrigation may be necessary in an arid zone, while in a humid region, pesticide treatments against fungal pathogens may be required. Crop management (i.e. the human intervention to the agro-ecosystem) obviously influences the yield. In general a more intensive production (e.g. higher amounts of fertilisers, more intensive soil cultivation, more irrigation or more frequent pesticide applications) will lead to a higher yield, but the relationship is usually not linear. Furthermore, there exists an interaction with the environment. In general under optimal pedo-climatic conditions a higher yield will be achieved with at the same intensity compared to a less optimal environment.

The environmental impacts are influenced by all three elements: obviously, the yield will have a dominant effect in a product LCA, where we use 1 kg of product as the reference flow. The same holds for crop management; more intensive management will generally lead to higher impacts per area unit, but not necessarily per product unit. Here the result depends on the ratio of impact and yield and therefore intensive management can have higher, similar or lower impacts as compared to extensive management. The pedo-climatic conditions finally will influence the environmental impact either directly through effects on emissions (e.g. soil erosion, leaching, ammonia volatilisation) or indirectly through an influence on crop management and yield.

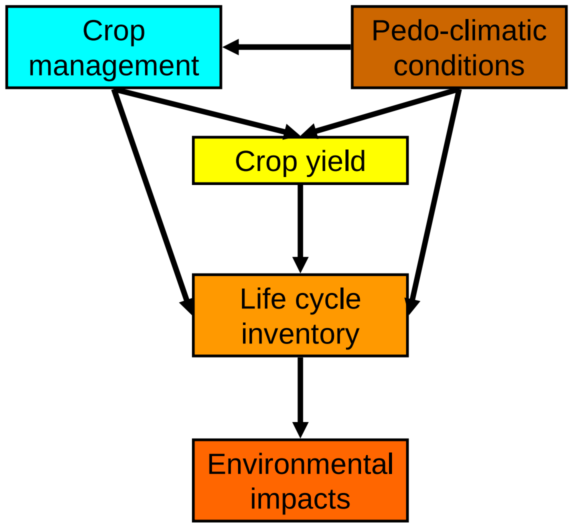

Fig. 2: Schematic representation of the relationships between pedo-climatic conditions, crop management, crop yield and the environmental impacts (per product unit).

Variability of yields

Let us consider first the variability of yields and production conditions. Relatively good data are regularly gathered on yield, and data on pedo-climatic conditions exist at least on a macro-level. Data on crop management are more difficult to obtain. Fig. 3 shows the distribution of wheat yields for all countries with a production area of at least 10’000 ha (i.e. minor producers were excluded). The 5-year averages for 2003-2007 vary between 106 kg/ha for Eritrea to 8785 kg/ha in Ireland (i.e. by a factor of 83)!

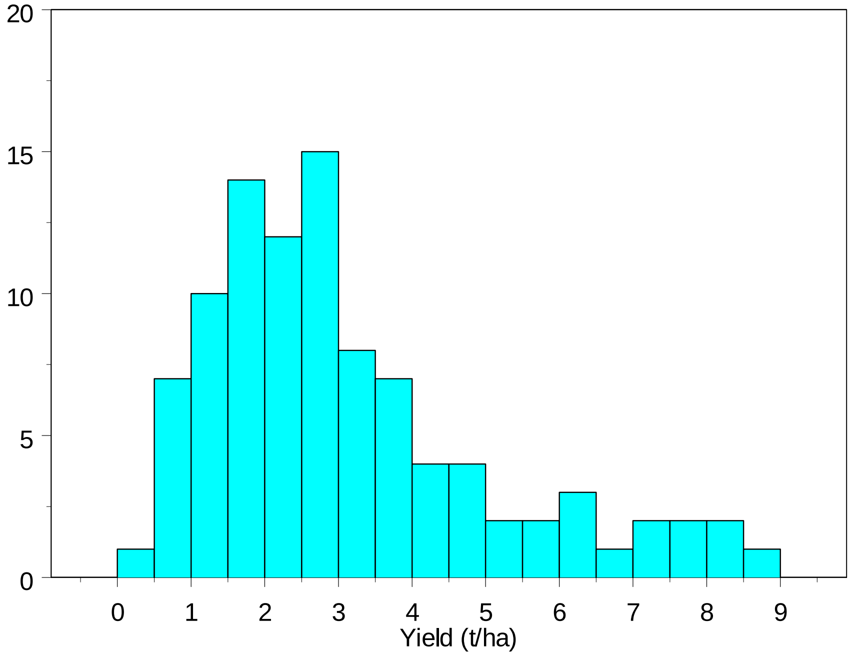

Fig. 3: Distribution of average wheat yields for the years 2003-2007 of 97 countries having a harvested area of at least 10’000 ha. The Y-axis shows the number of countries. Source: FAOSTAT (2008).

Variability of pedo-climatic conditions

But even within one country, the growing conditions may be very variable. This is illustrated by the example of the world largest wheat producer China. Winter wheat is grown in the warmer areas with higher precipitations, while spring wheat is reserved for cooler and drier climates (Fig. 4). The conditions of wheat production are very variable in terms of precipitations and soils (Fig. 5). We can expect that the environmental impacts per kg of produced wheat will be highly variable as well in function of the differences in climate and soil. In other words, an average inventory for Chinese wheat is of limited value; several inventories need to be defined in the different situations.

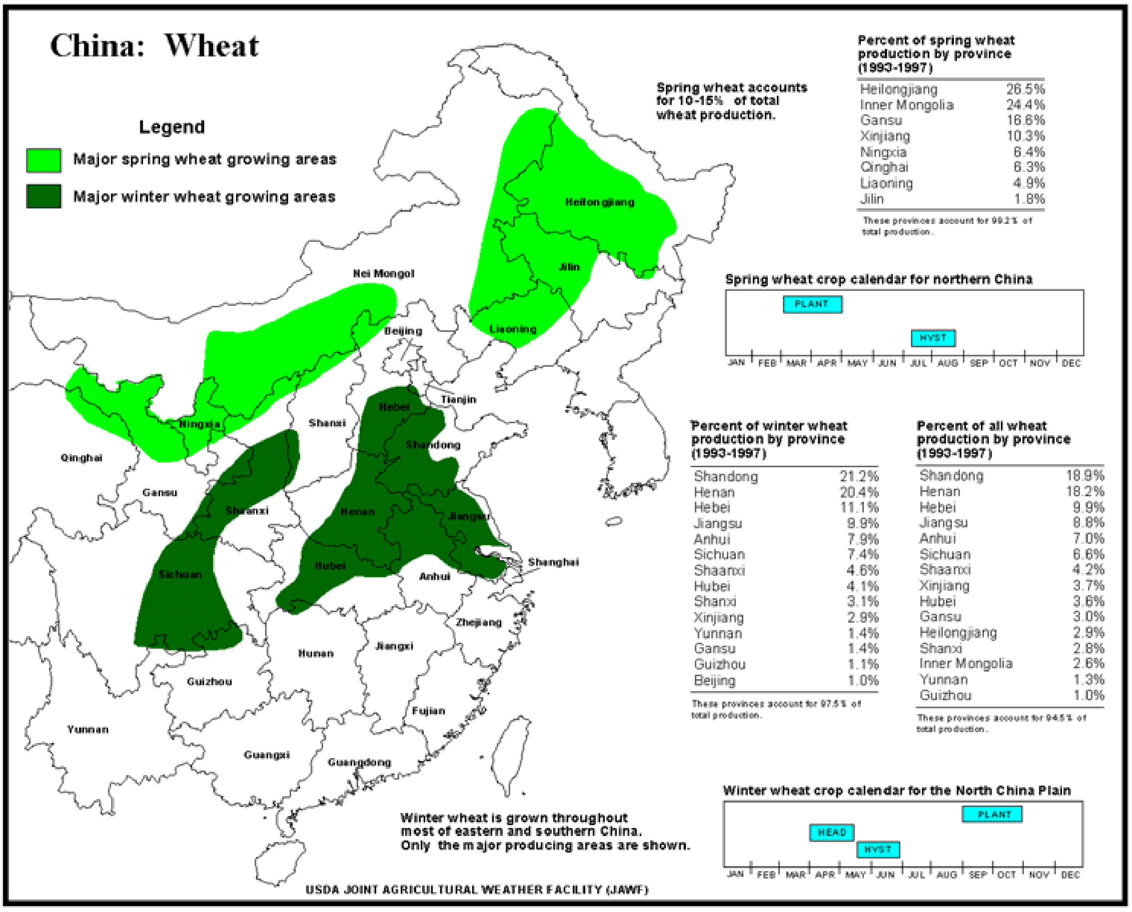

Fig. 4: Major wheat production regions in China. Source: IPNI (2008).

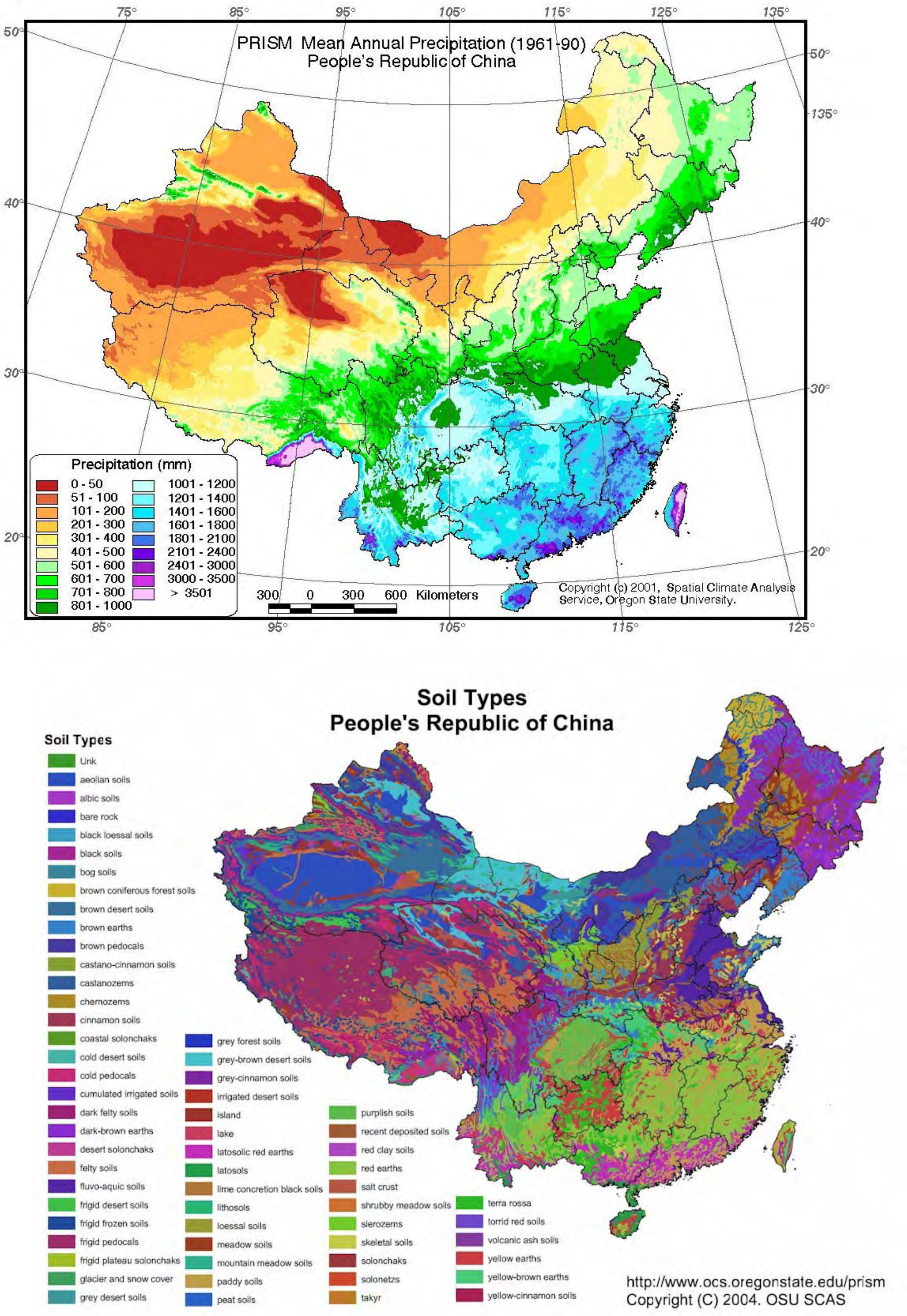

Fig. 5: Average precipitation and soil types in China. Source: Oregon State University (2008).

Crop management and environmental impacts

To see how changes in crop management are related to the environmental impacts, the relationships are analysed quantitatively for the example of wheat by means of a Monte-Carlo uncertainty analysis (see Nemecek *et al*., 2005). Since it is not meaningful to vary all parameters independently, four management areas were defined for yield, mechanisation, fertilisation and plant protection. For each management area, a variation factor was introduced:

- *Yield factor*: the yield is a key parameter for a product LCA.
- *Mechanisation factor*: the use of the machinery is important for many environmental impacts. This factor is multiplied with all machine work processes except those related to plant protection and fertilisation (see below).
- *Fertilisation factor* influences the quantities of fertilisers spread (mineral and organic) as well as the work processes related to spreading of fertilisers and direct field emissions of nutrients.
- *Plant protection factor* is multiplied with the quantities of pesticides applied and with the operations of the field sprayer.

Winter wheat from intensive integrated production in Switzerland was chosen as a baseline for this analysis (product LCA calculated per kg of wheat grains). The variability of the four factors was calculated from a pilot farm network (Zimmermann, 2003) for yield, mechanisation and fertilisation and from the farm accountancy data network for plant protection (Eggimann & Mollet, 2000). These four factors thus reflect the variability of the yields and input data.

Yield turned out to be the most important factor, which is strongly negatively correlated to all environmental impacts (Tab. 1). The negative correlation is a result of the division by yield in determining the impacts per kg product. The second important factor is fertilisation; as expected it has a high correlation to the use of the mineral resources P and K, to the nutrient-driven impacts acidification and eutrophication and to global warming and to a lesser extent to energy demand. The mechanisation is significantly correlated to energy demand and ozone formation, but the correlations are relatively weak. The plant protection factor is strongly correlated to the ecotoxicity and human toxicity.

Tab. 1: Correlation coefficients between the four variation factors and the different impacts per kg of winter wheat (Monte Carlo analysis with 200 runs). Shadowed cells mean significant correlations.

Variation factor	Energy demand	Global warming pot.	Ozone formation	Resource P	Resource K	Eutrophication	Acidification	Aquat. ecotoxicity	Terr. ecotoxicity	Human toxicity
Yield	-0.81	-0.72	-0.84	-0.67	-0.67	-0.67	-0.72	-0.70	-0.60	-0.68
Mechanisation	0.24	0.11	0.37	0.05	0.05	0.06	0.12	0.13	0.10	0.13
Fertilisation	0.49	0.66	0.34	0.72	0.72	0.71	0.65	0.26	0.13	0.22
Plant protection	0.09	0.07	0.08	0.06	0.06	0.06	0.07	0.64	0.76	0.67

We can conclude from this analysis that the knowledge of the yield is a key factor for a product LCA and that a good knowledge and understanding of processes related to fertilisation are crucial for a crop LCA. Good data on pesticides applied are indispensable for the impacts ecotoxicity and human toxicity, while even less precise data or approximations are acceptable for the use of the machinery.

Extrapolation to other geographical areas

The relationship between crop management, yield and environmental impacts are not simple. The closest relationship exists between yield (respectively the inverse of yield) and the impacts related to land occupation. Land occupation is closely related to the inverse of the yield, the duration of the vegetation period, land needed for seed production and other agricultural inputs make up the difference. The result of the method EcoIndicator ’99 is closely correlated to land transformation and occupation. Fig. 6 shows the linear regression between the inverse of yield and the EcoIndicator ’99 points. In the situation, where no land transformation occurs (which is the regular case for most agricultural systems), the link to the yield is quite strong. Simply spoken, the lower the yield, the higher the impact per kg of product.

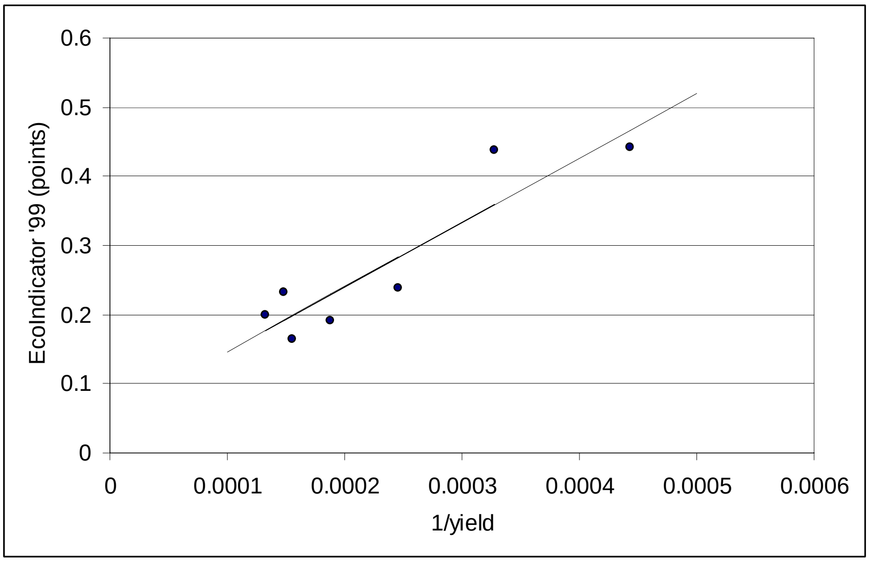

Fig. 6: Linear regression between the inverse of the yield (in kg/ha) and the EcoIndicator ’99 points wheat grains for six different wheat inventories from ecoinvent data V2.01 ($r^2 = 0.84$).

In these cases, a relatively simple correction could be applied to extrapolate impacts from a known situation 1 (with existing inventory) to a new situation 2:

$$I_2 = I_1 * Y_1/Y_2$$

where I_1 and I_2 is the impact in situation 1 and 2, respectively and Y_2 and Y_1 are the yields in these two situations.

For the other impacts, no such simple correction can be used. A few hints however can be given:

- Photochemical ozone formation is caused mainly by combustion processes. The amount of Diesel used by tractors could be used as an indicator.
- Cumulated energy demand (CED) is determined mainly by two inputs: Diesel and nitrogen fertiliser. Fig. 7 shows that a correlation between the nitrogen fertiliser input and energy demand exists. In irrigated agriculture, the process of irrigation is another important issue to consider, which significantly increases the CED. The outlier is the dataset from Castilla y León in Spain, a region with relatively dry summers and quite low yields of about 3 t/ha only.
- Global warming potential heavily depends on Diesel und nitrogen fertiliser use; however the relationships are more complex including direct field and farm emissions of nitrous oxide and methane as well as induced emissions of nitrous oxide from other nitrogen loss paths.
- Ecotoxicity and human toxicity are partly or mostly determined by applications of pesticides. However, the result depends on the method chosen and a thorough analysis of the applied active ingredients is required.
- For the use of water resources, obviously irrigation must be included.

Pedo-climatic conditions and the topography influence the direct field emissions through influences on processes like leaching, run-off, erosion, volatilisation, nitrogen mineralisation, nitrification, denitrification, C-sequestration and humus decomposition.

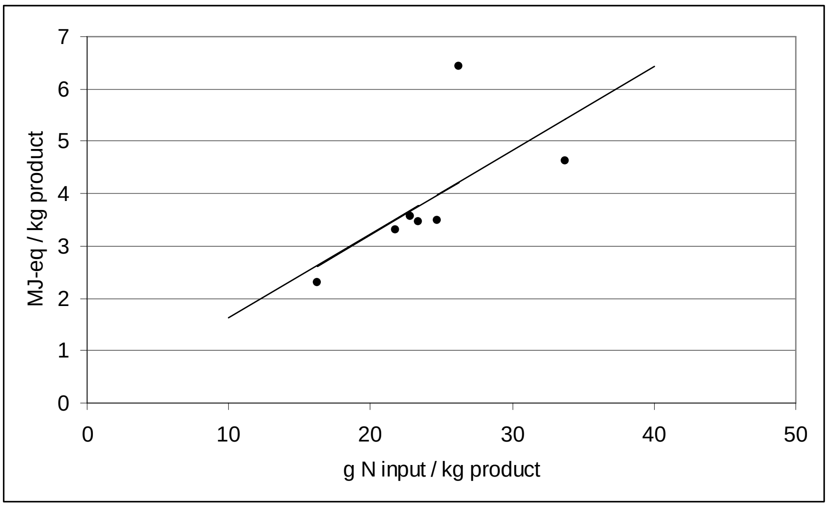

Fig. 7: Linear regression between N-input kg N per kg yield and non renewable energy demand MJ-eq. per kg wheat grains for six different wheat inventories from ecoinvent data V2.01 (r^2 = 0.42, after exclusion of the outlier r^2 = 0.95).

If a crop LCI is available in the ecoinvent database, but not for the considered country or a region, the question raises how to proceed? Several options exist. The best solution is to collect detailed inventory data specific for the situation under study and to establish a new inventory according to the ecoinvent rules. However, this is often not feasible for lack of resources, or because the dataset is not so important for the considered system. To extrapolate a new crop inventory from existing inventories, we can

- use an existing ecoinvent inventory with the most similar site conditions (climate, soil, topography) or management data (the above mentioned criteria help to decide, what “similar” means in the context of crop LCIs),
- apply a correction factor for the differences in yields and production means (as shown by the example above)
- combine two or more existing inventories by interpolating between them (for example 70% wheat Spain and 30% wheat France).

Conclusions and outlook

With a few exceptions, no simple extrapolation of LCI data from one situation to the other is possible. To make more reliable assessments in different geographical regions, we need to take the following actions:

- analyse of the variability of yields, pedo-climatic conditions and crop management,
- establish a relationship between the key factors and the environmental impacts,
- derive a method to extrapolate data in situations with poor data.

Based on this a better approximation of datasets should be possible.

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Comparison of air emissions for the construction of various greenhouses

H. Kowata[1], H. Moriyama[2], K. Hayashi[1] and H. Kato[1]
[1]National Agricultural Research Center, Tsukuba, 305-8666, Japan, [2]National Institute for Rural Engineering, Tsukuba, 305-8609, Japan, kowata@affrc.go.jp

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Abstract

Four types of greenhouses, i.e., pipe-framed greenhouse, multi-span greenhouse with circular arc roof, multi-span high-gutter greenhouse with truss beams and multi-span glasshouse with gable roof were chosen for comparative life cycle inventory analysis. The representative specifications for each type of greenhouse were selected. The data sets of unit environmental burden for the main structural materials of the selected types of greenhouses were created by the cumulative method with reference to the JLCA-LCA database. The emissions of environmental burden gases from the construction stage of greenhouses were estimated and normalized to a common functional unit per 1000 m^2 floor area of greenhouse. In the case of multi-span high-gutter houses with truss beams, the CO_2 emission from the manufacturing and transportation of structural materials is about 16600 kg-CO_2/1000 m^2. The base accounts for 23% of the total, steel materials for 67%, aluminium materials for 3% and transportation for 6%. Covering materials account for only 1%. NO_x and SO_x emissions were also analyzed. In comparing the CO_2 emission per unit floor area per year among the four types of greenhouses under the given conditions, the CO_2 emission of the arched roof house is 0.6 times that of the pipe house, 1.5 times that of the high-gutter house and 2.9 times that of the glasshouse.

Introduction

Many life cycle inventory (LCI) databases have been created worldwide (e.g., Curran, 2006). Typical examples are ecoinvent (Switzerland) and IVAM LCA Data (the Netherlands) and they include data concerning agricultural production systems. The creation of original and site-specific inventory data will play an important role in analyzing the current state of agricultural practices. However, there have been few reports on inventory data of greenhouses used for protected horticulture other than Russo (2005), and even ecoinvent does not address this issue. The recent trend toward larger greenhouses and a corresponding increase in their year-round use make the preparation of LCI data crucial. This is particularly true in Japan because of the introduction of new types of greenhouses and advanced cultural systems and increased emphasis on environmentally sustainable agriculture.

In this paper we focus on air emissions from the construction stage of greenhouses. The main aim of this research is to clarify which structural material has a larger environmental impact, by analyzing the inventory of structural materials according to the structural type regardless of the cultivation period or meteorological conditions, and thus to help decrease the environmental impact by improving the construction method. The load at the usage stage depends almost entirely on the heating load, and the amounts of input materials greatly depend on the local meteorological conditions, cultivation system and cultivation pattern. Therefore, if the construction stage is not separately analyzed when considering the load characteristic by type of greenhouse at the usage stage, it will not be possible to determine differences in load for various structures.

Analysis shows that for year-round tomato production in greenhouses in the Northern Kanto region in Japan, the CO_2 emission from structural materials is only about 3% of the CO_2 emission from crude petroleum used for heating. The ratio in Kyushu, which has a milder climate and so the heating load is smaller, is about 6%. According to LCA of greenhouse production, these relatively small values suggest that the CO_2 emission from structural materials can be considered negligible, and so inventory analysis of structural materials is not performed.

The objective of this research is to create an inventory database for structural materials of several types of greenhouses used for protected horticulture in Japan, and to compare the environmental impact arising from the structural materials used to build the greenhouses by using the LCA method. First, inventory data of structural materials used for various greenhouses in Japan were compiled. Then, LCI analysis was carried out for manufacturing and transporting structural materials of a modern greenhouse. In addition, the emissions of environmental burden gases resulting from structural materials of various greenhouses were estimated and compared.

Method

1. System boundary

Since many greenhouses are equipped with a heating system using fossil fuel to maintain the temperature necessary for cultivation in winter, the LCA result will be dominated by such fuel consumption. However, soaring crude oil prices now affect not only the oil price but also the cost of construction materials, include those for greenhouses. Therefore, the system boundary of the inventory analysis of this study was limited to the manufacturing and transportation of structural materials only. As shown in Fig. 1, the analyses in this study excluded the consumption of fuel and other inputs, as well as the management of waste plastics.

2. Types of greenhouses

Various types of greenhouses are used in Japan according to the cultivated crop, management scale, usage period, meteorological conditions, etc. Tab. 1 shows the main four types of typical greenhouses used to compile the inventory data. The pipe-framed greenhouse (hereafter referred to as "pipe house") is the most common greenhouse with the simplest structure. The multi-span greenhouse with circular arc roof (hereafter referred to as "arched roof house") is widely used for larger management scale. The multi-span high-gutter greenhouse with truss beams (hereafter referred to as "high-gutter house") is the most modern type of greenhouse suitable for the high-wire attraction method for long-term multistage cultivation. These three types of greenhouses are normally covered with plastic film such as polyvinyl chloride (PVC) or polyolefin (PO). The multi-span glasshouse with gable roof (hereafter referred to as "glasshouse") is framed by wider steel materials and more aluminium materials are used for fitting glass plates on the roof and side walls.

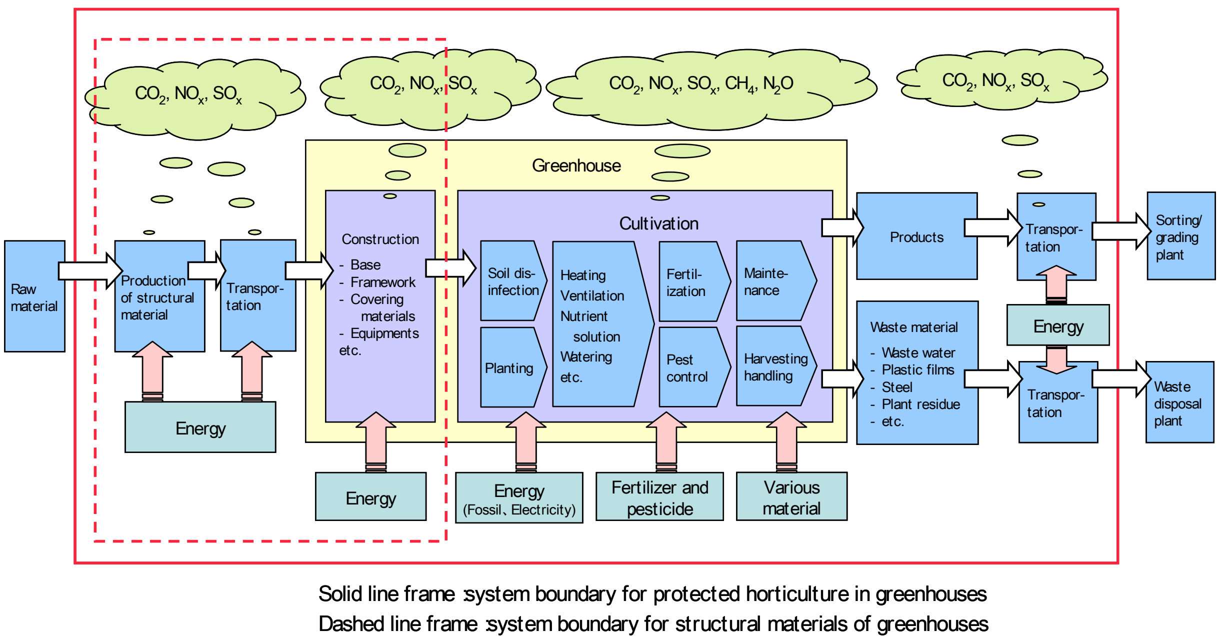
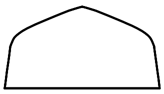
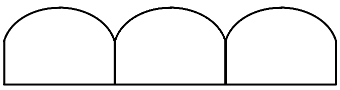
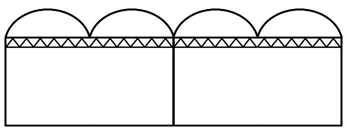
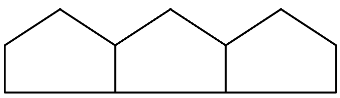


Fig. 1: System boundary for life cycle inventory analysis of structural materials of greenhouses.

3. Selection of structural materials

Because the specifications of greenhouses vary widely depending on the type, region and greenhouse manufacturer, common structural materials that are produced based on the Japan Industrial Standard (JIS) were mainly collected for compiling inventory data. In practice, the brochures of greenhouse manufacturers were reviewed, and series of structural materials for greenhouses of the same category were selected in order to cover several standard variations of each of the four types of greenhouses.

Tab. 1: Types of greenhouses and their primary specifications for comparative analysis.

Types of greenhouses	Representative specifications
Pipe-framed greenhouse (Pipe house) 	Floor dimensions: 6.0 m wide x 100 m long Floor area: 1000 m^2 Eave height: 1.75 m Base: steel, spiral anchor pile Arch frame: steel pipe, 25.4 mm-dia. x 1.2 mm-t Cross beam: steel pipe, 25.4 mm-dia. x 1.2 mm-t Covering material: polyvinyl chloride (PVC) film, 0.1 mm-t
Multi-span greenhouse with circular arc roof (Arched roof house) 	Floor dimensions: 6.0 m x 7 spans x 100 m long Floor area: 4200 m^2 Gutter height: 2.1 m Base: RC, simple column footing Principal rafter: steel pipe, 48.6 mm-dia. x 2.3 mm-t Cross beam: steel pipe, 19.1 mm-dia. x 1.2 mm-t Column: steel pipe, 48.6 mm-dia. x 2.3 mm-t Covering material: PO film, 0.075 mm-t
Multi-span high-gutter greenhouse with truss beam (High-gutter house) 	Floor dimensions: 9.0 m x 8 spans x 56 m long Floor area: 4160 m^2 Gutter height: 3.85 m Base: RC, simple column footing Roof arch: steel pipe, 31.8 mm-dia. x 1.6 mm-t Column: square steel tube, 100 mm x 50 mm x 3.2 mm-t Truss, square steel tube, 50 mm x 50 mm x 2.3 mm-t Covering material: polyolefin film, 0.15 mm-t
Multi-span glasshouse with gable roof (Glasshouse) 	Floor dimensions: 7.2m x 2 spans x 60.4 m long Floor area: 870 m^2 Gutter height: 2.1 m Base: RC, simple wall footing Principal rafter: steel H beam 100 mm x 50 mm x 5 mm x 7 mm Column: H-section beam, 100 mm x 50 mm x 5 mm x 7 mm Rafter: extruded aluminium Covering material: glass plate, 3 mm-t

The structure of the greenhouse was divided into several parts, namely the base, the framework and the covering materials, and then original materials were determined. For example, reinforced concrete used for the base was divided into Portland cement, gravel and reinforcing steel. As for steel materials of the framework, the types of materials were classified according to JIS.

4. Creation of inventory data

To collect inventory data (emission intensity of CO_2, NO_x and SO_x) from background data, the JLCA-LCA database was consulted in principle. As for the transportation of a structural material, when the cost of transportation was known but the fuel consumption and transporting distance were unknown, the emission intensity on a producer price basis given in the 3EID database (National Institute for Environmental Studies, 2002) was used.

5. Inventory analysis

5.1 Analysis for high-gutter house

A high-gutter house with 4160 m^2 floor area was chosen for the LCI analysis. The blueprint and preliminary estimate of an actual greenhouse were obtained from the owner, and the gross weight of each item of the inventory (weight, volume or length) was listed. The emissions of environmental burden gases (CO_2, NO_x and SO_x) from the manufacturing process for each kind of material were calculated. The unit floor area of a greenhouse was defined as a functional unit, and the amount of emission was expressed in units of kg-gas/1000 m^2. As the material transportation cost was known, the emission from the transportation stage was estimated using the 3EID database. However, the weight of aluminium materials was not known as they were not standard products. Therefore, the weight of aluminium materials was estimated by using the price ratio of the amount of aluminium and steel materials and by using the ratio of price per unit weight of steel and aluminium calculated from data in input-output tables of Japan.

5.2 Comparative analysis for four greenhouses

The main specifications for the selected four types of greenhouse were as shown in Tab. 1. The pipe house was a single-span structure and the base was a steel spiral anchor pile. The actual structural material lists for the pipe house and arched roof house were drawn up by using a greenhouse design support system (a sort of CAD system) imitating actual greenhouses. In this study, all four greenhouses were assumed to be single-covered structures. Emissions from the transportation stage and the construction stage were excluded since the required data were not available for every type of greenhouse. The amounts of materials per unit floor area for each type of greenhouse are listed in Tab. 2.

For analyzing the emissions per year according to the life of materials, the life time of each material was set as listed in Tab. 3. The values for the base and framework were quoted from the Japan Greenhouse Horticulture Association (1997).

Tab. 2: Amounts of materials for four different types of greenhouses (Unit: kg/1000 m^2)

Type of greenhouse	Base		Framework		Covering material		
	Concrete	Steel	Steel	Aluminium	PVC	PO	Glass
Pipe house		121	3898		434		
Arched roof house	5362	28	5499			101	
High-gutter house	14858	388	9174	621		279	
Glasshouse	67747	954	11818	2549			10233

Tab. 3: Assumptions for life of structural materials of greenhouses

Type of greenhouse	Base and framework	Covering material
Pipe house	10 years	1 year
Arched roof house	15 years	5 years
High-gutter house	15 years	5 years
Glasshouse	20 years	20 years

Results and discussion

Environmental burden of high-gutter house

CO_2 emission

Fig. 2 shows the result of the LCI analysis of the high-gutter house. Total CO_2 emission per 1000 m^2 is about 16600 kg-CO_2/1000 m^2. The base accounts for 23% of the total, steel materials for 67%, aluminium materials for 3% and transportation for 6%. Covering materials account for only 1%. The reason why the ratio of the covering material is unexpectedly small is that the total weight of polyolefin film is only about 3% of the total.

NO_x and SO_x emissions

The NO_x emission is 26.5 kg-NO_x/1000 m^2, of which transportation accounts for 25%. The SO_x emission is 9.9 kg-SO_x/1000 m^2, of which steel material accounts for about 80%, and transportation about 10% which is not negligible.

Discussion

The average weight of the framework per unit floor area is about 10 kg/m^2, while that of the base is about 15 kg/m^2. Because the wind resistance of the high-gutter house depends on the weight of the base, the base weight is much greater than that of the arched roof house. To decrease the environmental impact of this type of construction, a construction method that uses less concrete is needed.

Regarding ordinary LCA of greenhouse production, the air emission per unit floor area is not so important because the life time of materials is not taken into account. In general, the life time of each material is defined by some rules, but the actual life time of greenhouses differs greatly depending on climatic conditions, cultivation system, cultivation period and so on. Therefore, if we focus on environmental evaluation of greenhouse construction only, the initial values of environmental burdens from the construction stage are easier to understand, which helps to identify problems and make improvements. Furthermore, values in units per 1000 m^2 are directly related to the cost of constructing greenhouses, and so manufacturers can easily compare the effect of improving the construction method regardless of the cultivation system used by the farmer.

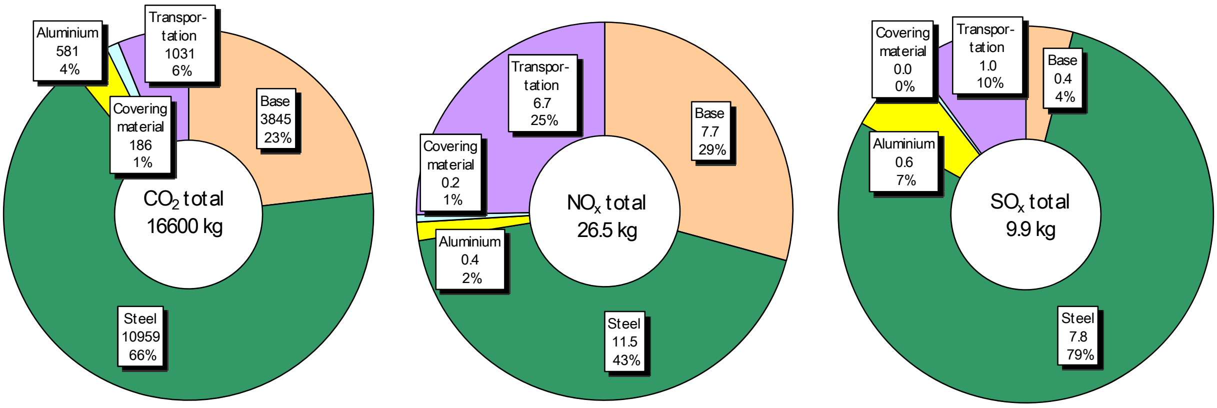


Fig. 2: Emission of environmental burden gases per 1000 m^2 from structural materials of a multi-span high-gutter greenhouse with truss beams.

Comparison of emission of environmental burden gases among various greenhouses

CO_2 emission and component ratio

Fig. 3 shows the CO_2 emission per 1000 m^2 for structural materials of various greenhouses and Fig. 4 presents the composition ratio of CO_2 emission of each greenhouse. The CO_2 emission of the pipe house is about 3000 kg-CO_2/1000 m^2, with the framework accounting for a relatively large proportion of about 90%. The covering material accounts for about 10%, which is characteristically larger than that of other plastic-covered houses. The CO_2 emission of the arched roof house is about 6000 kg-CO_2/1000 m^2, of which the base accounts for about 20% and the framework for 80%. The CO_2 emission of the high-gutter house is around 15000 kg-CO_2/1000 m^2 (excluding the transportation stage), with the base accounting for a slightly larger percentage than that of the arched roof house. The composition ratio of covering materials is only 1% or more for both the arched roof house and high-gutter house.

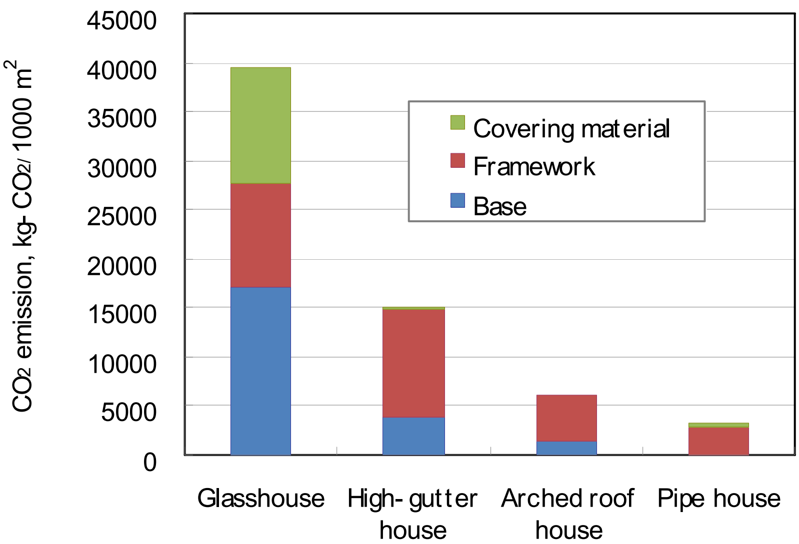


Fig. 3 CO_2 emission per 1000 m^2 for structural materials of various greenhouses

The CO_2 emission of the glasshouse is the largest with around 39000 kg-CO_2/1000 m^2, and the percentages in descending order are the base at 43%, covering material (glass plates) 30%, and framework 27%. In contrast to the other three types of greenhouses, the percentages of the base and covering material are considerably large. Comparing the total CO_2 emission among the four greenhouses relative to the value for the pipe house, the emission is 1.9 times for the arched roof house, 4.7 times for the high-gutter house and 12.4 times for the glasshouse.

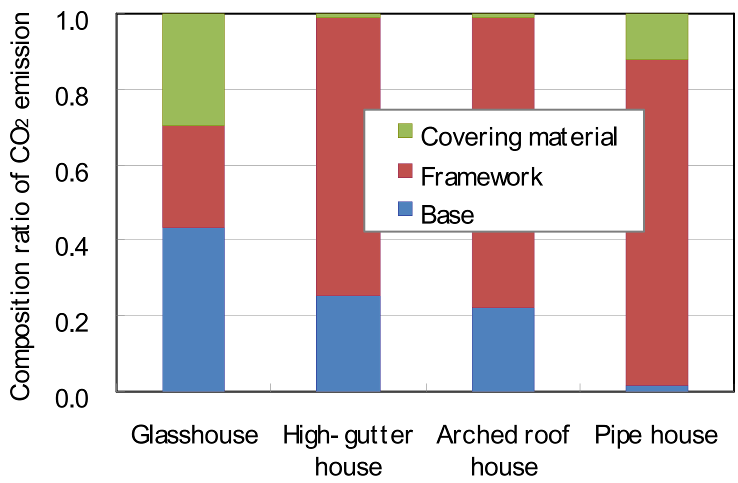


Fig. 4: Composition of CO_2 emission

NO_x and SO_x emissions

Fig. 5 shows NO_x and SO_x emissions per 1000 m^2 floor area of various greenhouses. The values of NO_x emission vary from 3 kg-NO_x/1000 m^2 for the pipe house to 55 kg-NO_2/1000 m^2 for the glasshouse. The base accounts for a greater percentage of NO_x emission than CO_2 emission. For the arched roof house and high-gutter house, the base and framework account for approximately 40% and 60%, respectively. As for the glasshouse, the percentage of covering material is greater than that of the framework. The SO_x emission is 2 to 8 kg-SO_x/1000 m^2, the majority of which is due to the framework, and the emission from the high-gutter house is slightly higher than that of the glasshouse.

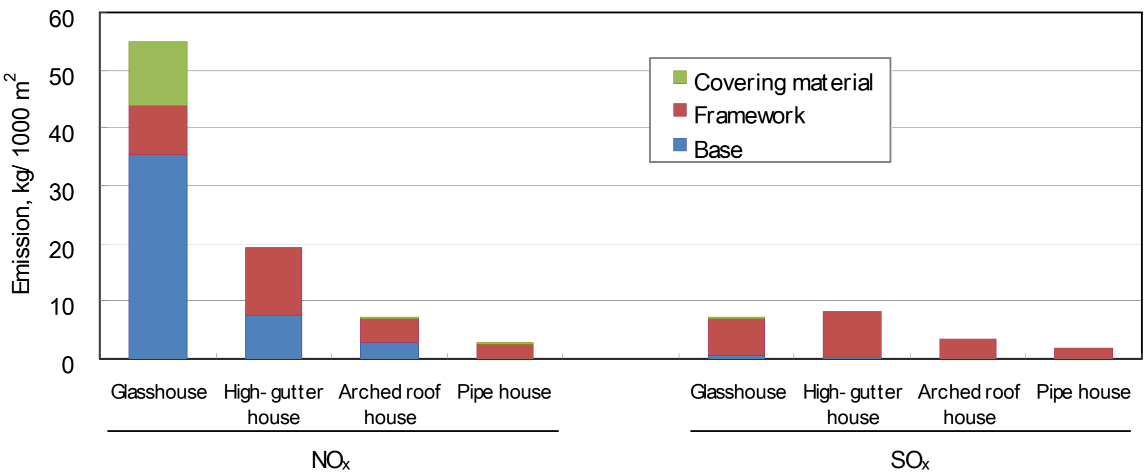


Fig. 5: NO_x and SO_x emissions per 1000 m^2 floor area of various greenhouses.

CO_2 emission per year

Fig. 6 shows the CO_2 emission per 1000 m^2 floor area per year from the four types of greenhouses. Considering the life of each material, the CO_2 emission per year from the pipe house is about 670 kg-CO_2/(1000 $m^2 \cdot y$), which is much greater than that from the arched roof house with about 410 kg-CO_2/(1000 $m^2 \cdot y$). The main reason for this inversion phenomenon is that the life time of PVC film used for pipe houses is one fifth of that for PO film. The CO_2 emission from the high-gutter house is about 1020 kg-CO_2/(1000 m^2·y), and that from the glasshouse is 1970 kg-CO_2/(1000 m^2·y). Compared with Fig. 3, the ratio of the smallest value to the largest value reduces from 12.4 times to 4.9 times. Relative to the CO_2 emission from pipe houses, that from arched roof houses is 0.6 times smaller, that from high-gutter houses is 1.5 times greater, and that from glasshouses is 2.9 times greater.

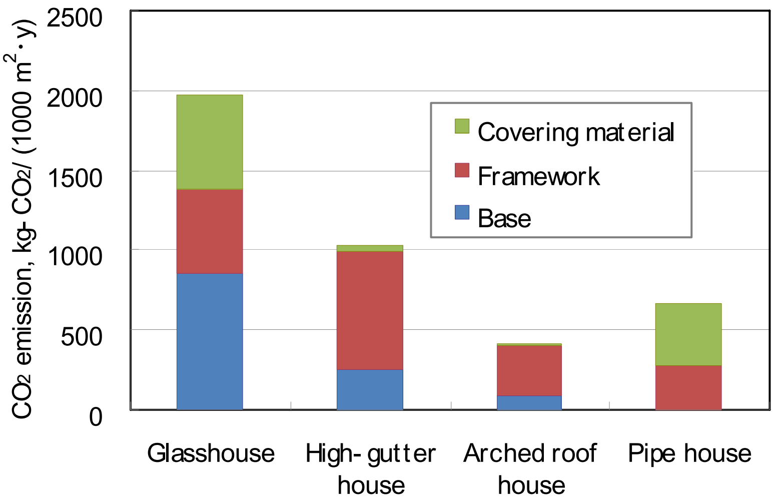


Fig. 6: CO_2 emission per 1000 m^2 floor area per year from four types of greenhouses.

Discussion

According to Fig. 3 and Fig. 5, the CO_2 and NO_x emissions from the base account for significant proportions of the total. Therefore, a simpler method of constructing the base would effectively reduce the environmental burden from greenhouses. For this reason, a company has recently developed a new technology that uses spiral steel piles for the base instead of concrete. Because the life of the film influences the emission per year, the emissions of CO_2 and other gases can be reduced by changing the covering material of the pipe house to materials with higher durability. To conduct an LCA of protected horticultural production under particular scenarios, it is necessary to compile inventory data for heating equipment, management of waste plastics, fertilizer and pesticide, etc.

Conclusion

The inventory analysis for the construction of several types of greenhouses showed that the CO_2 emission per 1000 m^2 per year varies widely from 410 to 1970 kg-CO_2/(1000 m^2·y) depending on the type of greenhouse. The emission of environmental burden gases per year depends mainly on the amount of concrete used and the life of the plastic covering film. Therefore, to reduce gas emissions, it is necessary to introduce concrete-free construction technology and long-life covering materials. Based on this study, it is expected that a CAD system would help analyze the inventories of structural materials of greenhouses with various specifications.

Acknowledgements

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Creating Life Cycle Inventories using systems modelling to compare agricultural production alternatives

E.Audsley[1] and A.G.Williams[1]
[1] Natural Resources Management Centre, Cranfield University, Bedford, MK43 0AL, United Kingdom, e.audsley@cranfield.ac.uk.

Keywords: agriculture, arable, dairy cows, wheat, system modelling

Abstract

Different ways are suggested to reduce the environmental burdens of agricultural production, including reducing fertiliser use, increasing yields or even producing the food elsewhere. Some are hypothetical or have limited experimental data. We need to know how environmental impacts and land use are changed by long-term widespread use of alternative systems of primary agricultural production for a nation's food. This has become more important as our sources and seasonal consumption of food have diversified and land use for biofuels is increasing. The Cranfield University LCI approach (www.agrilca.org) is to use models of systems and processes. The approach is described using bread wheat and milk as examples. In arable production, a long-term analysis of emissions and yields in rotations ensures that short-term effects are not presented as long-term solutions. Inputs and outputs are also correlated with soil texture. Animal production models are used that define industry breeding structures and link outputs to nutritional demand, fertility, productivity and manure (and enteric methane) production. System and process modelling considerably enhances the LCA of food production and provides a highly interactive framework for the analysis. It also highlights data gaps and limits to our knowledge. Within the framework, one can study alternative agricultural techniques and, using the results, examine alternative food consumption.

Introduction

Environmental Life Cycle Assessment (LCA) is used to calculate the burdens of producing one unit of a food commodity, such as 1 kg of wheat or meat, or a litre of milk, and getting it to a common point, in this case the farm gate. Some commodities may also be defined by a qualitative property or season. The burdens result from the use of resources such as fossil energy, and emissions to the environment, such as nitrate, ammonia, nitrous oxide and fossil CO_2. A typical result from LCA is that it requires, say, x MJ per kg wheat with a dry matter concentration of 86%. This result is perfectly reasonable and useful, but a question that almost always follows is: what happens if the production system changes? To answer that question reliably requires the production system to be modelled so that changes can be calculated. This approach has been extensively implemented in the Cranfield LCA models of agricultural and horticultural commodity production. The system modelling includes:

- structural models of breeding and replacement in the livestock sector
- nutrient flow models within the livestock and arable sectors
- process models applying to soils, crops, post-harvest activities and animal production.

In the following sections examples are illustrated for non-organic bread wheat and liquid milk production.

Approach

Crop production

Crop production is mainly arable, although grass is also a specialist crop. A fundamental tenet of cropping is that the main nutrients of nitrogen (N), phosphorus (P) and potassium (K) balance on a long term basis.

Nitrogen

N is the most important element to analyse given:

- its vital role in limiting or enabling yield and providing protein
- the high energy cost of manufacturing synthetic N
- its vulnerability in the soil, causing emissions to the environment including ammonia, nitrate and nitrous oxide, which contribute to acidification, eutrophication and global warming.

Leaching losses were calculated with the SUNDIAL model that was developed at Rothamsted Research (Smith *et al*., 1996). It simulates N uptake by crops and N turnover in soils and was initially calibrated against the long term experiments on the Broadbalk plots at Rothamsted. Rotations were set up with a variety of representative crops and the model was run until it reached steady state. This is illustrated by the fact that the organic N pool remaining constant at the start and end of a rotation, although it may change within the rotation (Fig. 1). The levels of fertiliser, soil type and rainfall were increased and decreased to investigate the effects in yield and N leaching. Meta-modelling was then used to interpret these effects and produce simple expressions to relate changes in N supply to leached N.

Yield and quality response to N supply was also traced back to experimental data from the Broadbalk plots. This of great importance as our crop yields are based on this philosophy. The Broadbalk yield curves follow the typical curve of the rest of agriculture (Fig. 2), but the main difference is the intercept when applied N reaches zero. In the long term Broadbalk experiments, when fertiliser N is zero, the crop N supply is effectively limited to atmospheric deposition and free-living N fixing soil bacteria. In this situation, N transfers between crops in successive years cancel out. The wheat yield is much lower than what could occur on a commercial farm in which N fertiliser was arbitrarily cut to zero for one year. In the latter case, soil pools can be mobilised to liberate crop-available N, but clearly not forever. If no more N is applied, this process would continue until a new soil equilibrium was established and the yield would fall to that of Broadbalk (given adjustments for rainfall and soil texture etc). In crops like wheat, the grain protein is also affected by N supply and, for a given wheat variety, grain protein concentration falls with N supply.

SUNDIAL calculates total denitrification, but does not separate N_2O from N_2. N_2O emissions were thus estimated using the IPCC (2001) methodology at the Tier 2 level. This has now been updated to the IPCC (2007) methodology. This relates direct N_2O emissions to the soil N supply from atmospheric deposition, N fertilisers and manures. Secondary emissions are also estimated from leached nitrate and ammonia (Tab. 1). It is a relatively simplistic approach as fixed emission factors are used irrespective of factors like timing of application, type of fertiliser or rainfall. IPCC also provides a simplistic approach for calculating nitrate leaching (30% of applied N), but we used that derived from SUNDIAL.

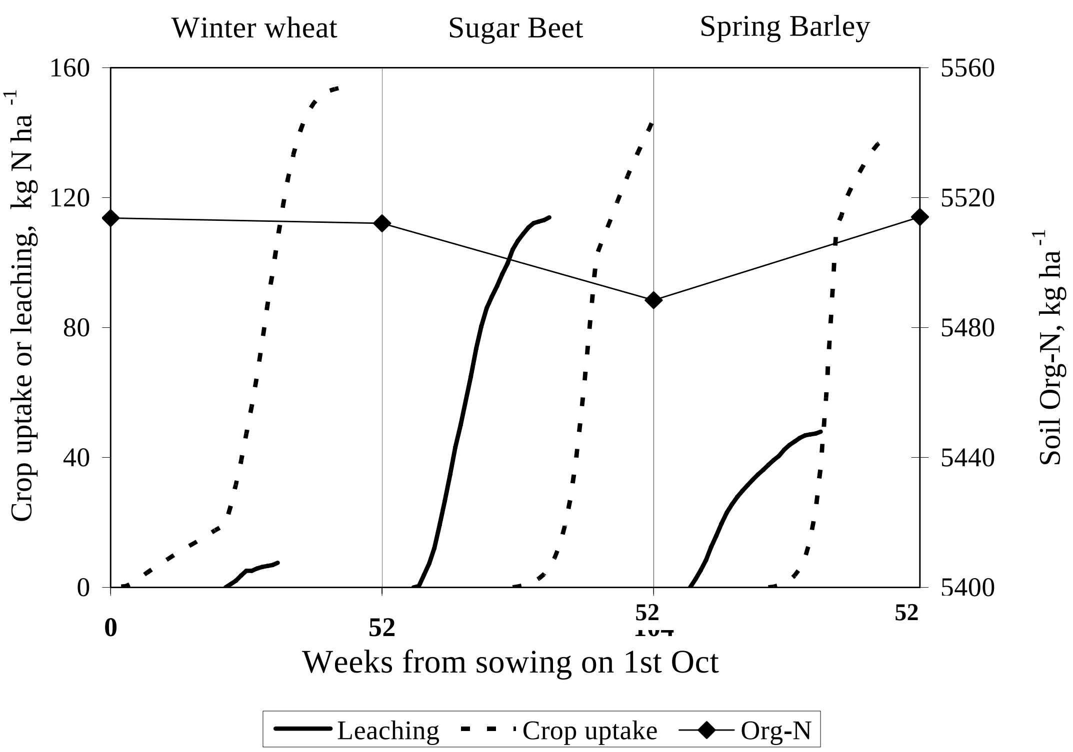

Fig. 1: Example of crop-soil simulation using SUNDIAL for a three crop rotation in steady state showing cumulative N accumulation in annual crops, cumulative leaching and soil organic N at the end of each crop season.

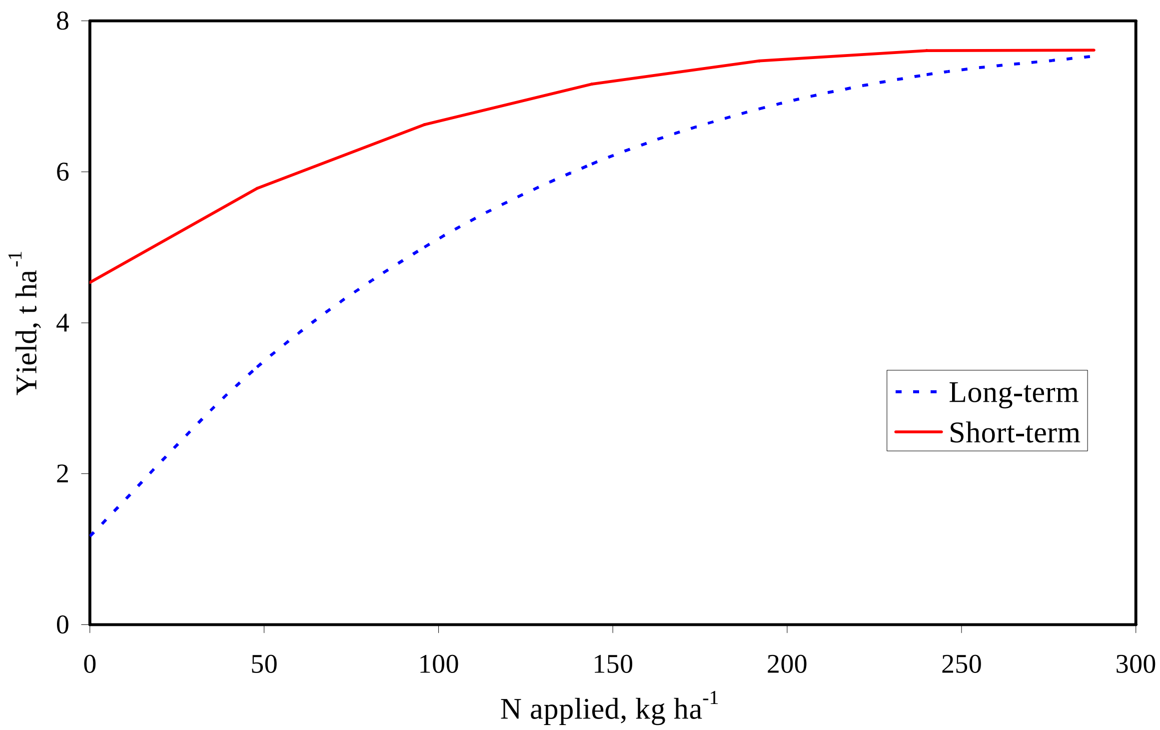

Fig. 2: Crop yield response curves based on long and short term cultivations.

Tab. 1: Emission factors used for calculating N_2O emissions using the IPCC 2007 methodology.

Source of N	Proportion of N emitted as N_2O-N
Direct emissions	
Fertiliser	0.01
Manure	0.01
Arable returns	0.01
Atmospheric deposition	0.01
Indirect emissions	
Ammonia and NO_x volatilisation (e.g. from fertilisers, manure or combustion)	0.01
Leached nitrate	0.0075

The modelling thus links N supply to yield and both denitrification and leaching losses. For bread wheat, as N supply increases, energy inputs per ha increase linearly because of fertiliser manufacture energy (Fig. 3). Total yield increases following a linear exponential curve, but because grain protein concentration is affected the proportion that reaches bread making quality is radically lower with low N supply rates. The balance of the yield still qualifies as feed wheat, of course. Note that it is possible to change to a variety which gives a higher protein concentration with lower fertiliser, but the yield is still lower.

The overall effects of these interactions were combined with other impacts (Fig. 4). This suggests that there could be an environmental optimum for reducing N supply to about 75% of its current norm for bread wheat with respect to energy use and GWP. One limit though is land occupation, which increases rapidly with reducing N application. In practice, an alternative solution is needed, e.g. developing bread wheat varieties that can function with lower N supplies without loss of yield or changing the bread wheat specification to accept types of wheat that are not currently considered to be suitable. This would require a public acceptance of other qualities of bread.

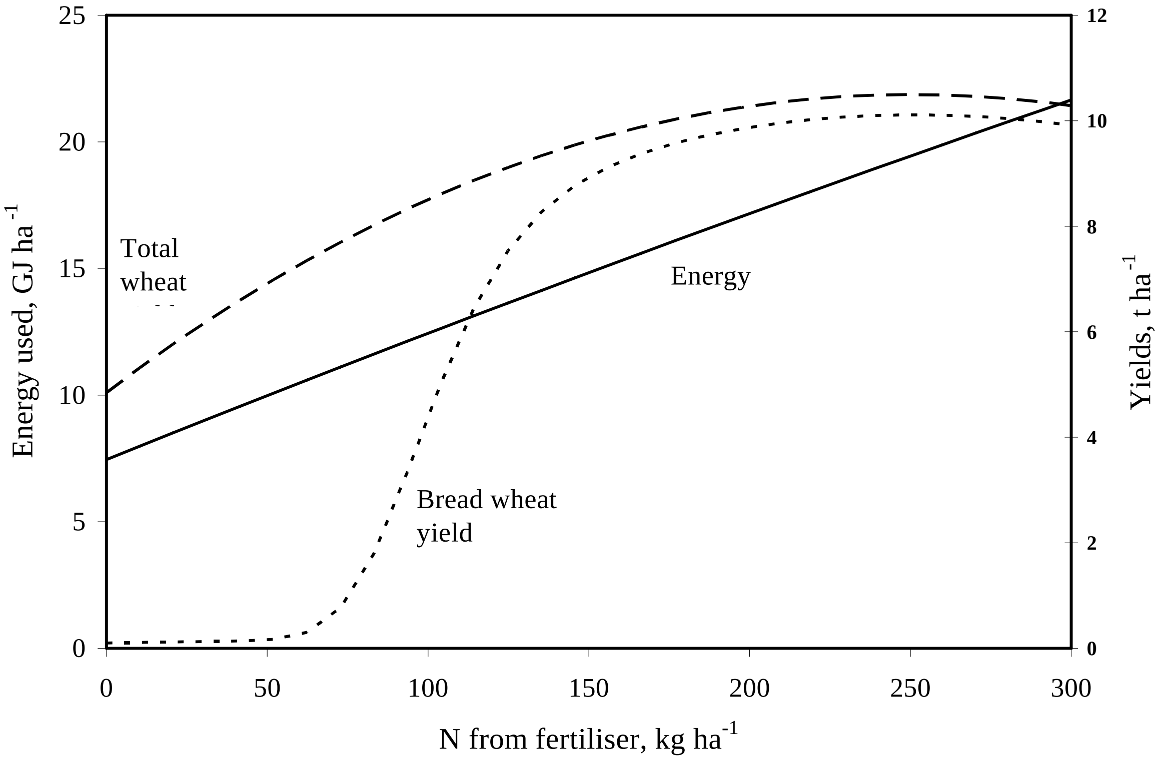

Fig. 3: Effects of fertiliser application rate on energy use per ha and total yields of wheat and bread wheat. The yields are net, i.e. excluding the seed rate.

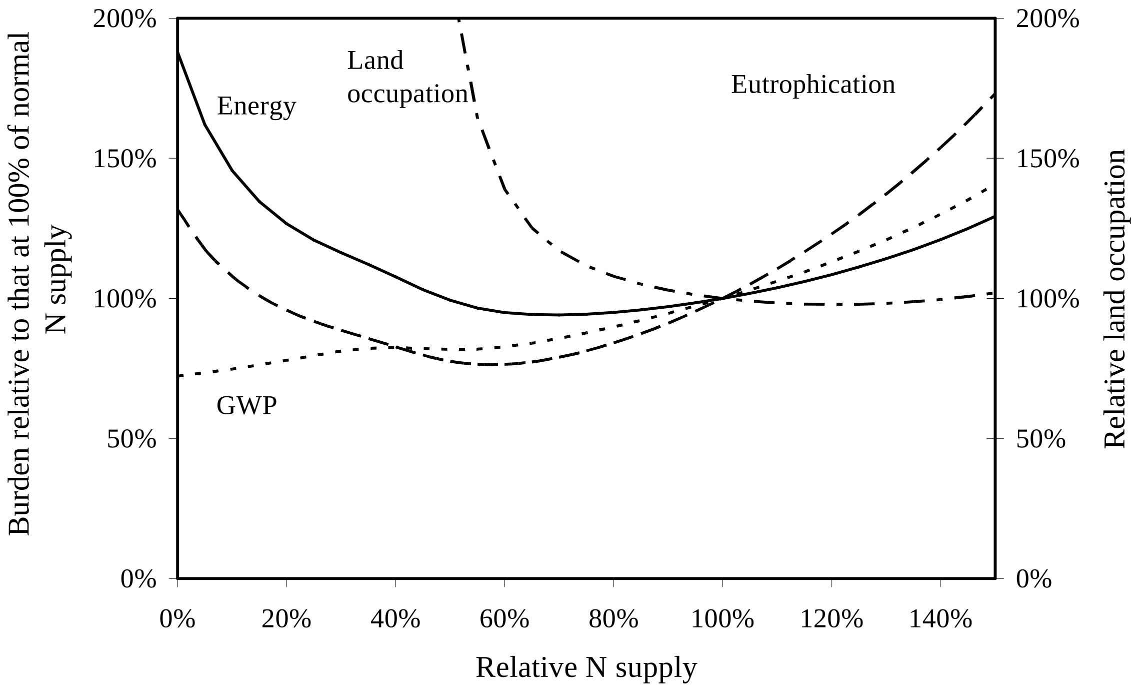

Fig. 4: Effects of changing synthetic N fertiliser rate on energy use, global warming potential, eutrophication potential and land occupation for producing 1 t bread wheat. All values are normalised against 100% fertiliser application being 220 kg N ha^{-1}.

Phosphorus and potassium

A simpler approach can be adopted for P and K as they are less mobile in the soil. In the long term, supply must equal offtake and losses to the environment. Losses to the environment were simplified to a mean of 1 kg ha^{-1} for each of P and K. An important variable to consider with cereal crops is whether straw is removed or not. This has most effect on K (Tab. 2). The choice does not exist for almost all other crops, in which straw or haulm incorporation is the norm.

Tab. 2: Removal main plant nutrients (as elements, kg ha^{-1}) in typical bread crop with either straw incorporation or removal.

	Straw Incorporated	**Straw Removed**
N	150	180
P	21	24
K	30	64

Soil and rainfall

In our use of SUNDIAL, a national distribution of soil texture and rainfall was derived, combining the distribution of soil textures in 5 km grid squares in the National Soil Resources Institute's inventory with long term rainfall from the UK Meteorological Office. In addition to the effects on leaching and denitrification (Tab. 3), energy use for cultivation and yield are also affected. Heavier soils require more energy, but support higher yields.

Tab. 3: Effects of rainfall and soil texture on losses of N by leaching and denitrification and energy use for crop establishment and crop yield.

Soil texture	**Rainfall level**	**National proportion, %**	**Leaching, kg N ha^{-1}**	**Total denitrification, kg N ha^{-1}**	**Energy for crop establishment, MJ ha^{-1}**	**Yield, t ha^{-1}**
Clay	dry	13	26	65	4300	8.5
	mid	12	32	60		
	wet	8	35	56		
Loam	dry	19	27	65	3200	7.3
	mid	17	33	60		
	wet	12	36	56		
Sand	dry	7	46	45	2600	6.0
	mid	6	45	46		
	wet	5	47	45		

Animal production

Animal production breeding structures are modelled with linear equations that represent the input-output relationships of each component, such as the ewe lambs coming from hill farms. The mathematics has been described in detail by Sandars *et al.* (2006). In summary, the components are linked such that changes in the proportions or output of any sector of an animal production system generate responses in the proportions of other sectors in order to continue to produce 1 tonne of meat or 1,000 litres milk.

Within each animal production system, the inputs and outputs are linked and milk production is described as an example. Nine milk production sub-systems are defined, based on three production levels (low, medium and high), and for spring, autumn calving and organic. Cow sizes tend to be higher for higher yielding cows, reflecting the trend for the higher yields to be delivered by Holsteins rather than Friesians. Each yield level is associated with characteristics, such as yield per lactation, lactation length, number of productive lactations and use of forage maize. The dietary needs are derived from the energy needs of lactation, together with maintenance and pregnancy. The energy for lactation depends on the volume and concentrations of milk fat, lactose and protein. The dietary protein requirement depends on similar factors.

The amount of feed is calculated from the forage types available (grazed grass, grass silage and maize silage), with concentrates used to supply the energy and protein needs of the cow within the appetite of the cow. The manure quantity and N concentration is derived from the difference between feed inputs and milk output. Enteric methane emissions are derived from the forage consumed, this being the source of methanogenic fermentation in the rumen.

Emissions of N from manure in various forms (e.g. NH_3, N_2O) are a function of the N excretion, with particular coefficients for different manure management systems. A manure model calculates all gaseous emissions from excretion, storage, spreading and land use. The manure is applied to grass or winter wheat and the long-term crop response to both readily-available and slow release N is calculated, together with long-term emissions of N. The yield response is quantified as if coming from a defined N source. The outcome is that emissions to the environment and the energy needs for manure management are debited against the livestock production system, and the crop yield response is credited as a fertiliser and land use saving to the livestock production system.

Some effects of changes in milk production systems are given (Tab. 4) and show how the modelling can illuminate features. If the yield of a typical cow is increased from 9000 litres, the proportion of concentrates in the diet must be increased because of the physical limit to intake of a cow. As concentrates take more energy to produce than forage, there is no energetic benefit even though the overheads of maintenance are reduced through more milk output per cow. GWP decreases slightly because the methanogenic supporting part of the diet is reduced. A typical response is to achieve the higher yield with a larger cow with larger appetite, e.g. breed substitution from Friesian to Holstein. This allows a higher proportion of forages in the diet, but reduces energy needs only slightly. A far better result for the environment could be obtained by breed improvement such that the efficiency of converting feed into milk energy is increased, e.g. by 8%. This allows the smaller cow to deliver the yield, with lower energy needs and emissions of GHG and ammoniacal N.

Tab. 4: Effects of changes in milk production systems on burdens of producing 1000 L milk.

	Lactation yield, L	Cow weight kg	Dietary concentrates, %	Primary energy, GJ m^{-3}	GWP, t CO_2 equiv. m^{-3}	NH_3-N, kg m^{-3}
Current	9,000	650	37	2.6	0.99	3.5
Yield up 15%	10,400	650	50	2.6	0.91	3.2
Breed change (typical)	10,400	720	40	2.5	0.95	3.3
Energy conversion up 8%	10,400	670	39	2.4	0.89	3.2

Discussion

Animal production systems are highly constrained and the results illustrate that the fundamental limit in all species and systems is feed conversion efficiency. Improving this is the key to reducing environmental burdens from livestock production. Achieving it is a major challenge for geneticists and allied animal sciences. Improving nutrient utilisation efficiency (especially N) is also critical in reducing the burdens of arable cropping.

The use of a model based approach allows changes in a system to be explored readily. By using process and structural modelling to underpin the analyses, we endeavour to ensure that all potential changes caused by modifying a production system are accounted for. It is important for the models themselves to be well founded on good data, whether experimental results, survey or activity data. A major part of the work in using such models a very thorough examination of data to ensure consistency and reliability. When data are lacking, a model or sub-model represents a hypothesis that needs testing. This leads to improvements in the modelling if the models are challenged with better data. Such a process of continual improvement is to be welcomed.

Conclusion

Model based Life Cycle Inventories of agricultural production have been produced for British systems. The principles apply anywhere, but models have been parameterised for Britain, particularly England and Wales. The use of the model based approach demonstrates the importance of linking systems and sub-systems together and does not allow short term practices to claim reduced burdens that are not justifiable. Improving nutrient utilisation by both crops and animals is of major importance in reducing the environmental burdens of agricultural production.

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LCA and carbon footprints in agro-food: From theory to implementation in the food industry

S. Deimling[1], P. Shonfield[2], U. Bos[3],
[1] PE INTERNATIONAL, Hauptstrasse 111-113, 70771 Leinfelden – Echterdingen, Germany
[2] PE North West Europe, Rectory Cottage, Rectory Gardens, Harthill, Sheffield S26 7YG, UK
[3] LBP University of Stuttgart, Hauptstrasse 113, 70771 Leinfelden – Echterdingen, Germany

s.deimling@pe-international.com

Keywords: LCA, Carbon-footprint, agricultural production, food industry, GaBi 4, PAS 2050

Abstract

There is increasing demand from stakeholders across the food supply chain to better understand environmental impacts associated with agriculturally-derived products. PE's Agrarian model allows robust assessment of these products accounting for the complexities of agricultural processes including crop rotations, carbon and nitrogen cycles, nitrate emissions, etc. The model can be used to assess different farming systems, crop types and growing locations.

Some challenges remain in terms of streamlining data collection, making modelling somewhat easier (for users not necessarily having agricultural background), and coping with competing national standards for carbon foot printing.

Introduction

Agriculture is vital to human welfare providing a range of products including food, textiles, fibres and fuels and giving employment and livelihood to millions of people across the globe. Without modern agricultural production systems society as we know it would not be able to function. Nevertheless, along with the many benefits derived from agriculture there are also significant associated environmental impacts. For example, the IPPC (2007) estimates that agriculture accounts for 10 - 12% of total global anthropogenic emissions of greenhouse gases (GHGs). This includes 60% of global emissions of nitrous oxide and 50% of global emissions of methane, which mainly occur in general animal production but especially in products derived from ruminant animals. Other impacts include eutrophication due to fertilizer use that can cause widespread damage to aquatic life, and deforestation resulting from demand for more farm land.

As a result there is increasing pressure from stakeholders in all parts of the supply chain to better understand the environmental impacts of agriculture. This ranges from the consumer who wants to make an informed choice about the food they buy, through to the retailer and food companies who seek to gain competitive advantage by offering greener products, and up to governments who are seeking to reduce national GHG emissions.

Reflecting this demand for more sustainable food production systems, the focus of agrarian modelling in LCA and CF has changed over the past five years, shifting from the debate on bio-fuels to an increasing requirement to better understand the sustainability, especially the carbon footprint, of food supply chains. LCA and CF approaches can help food companies to understand the impacts of their products.

In its 2006 report on the "Environmental Impacts of Food Production and Consumption" the Manchester Business School (2006) clearly states, that, to fill the identified gaps in measuring the environmental performance of food products, further LCA studies on food products and comparative studies of the environmental impacts of food production in different countries should be performed.

Recently, certain branches within the food industry (especially the dairy industry) are concentrating on the reduction of their products' environmental footprints [DSCF (2008); Sustainability Summit (2008)].

However, agrarian systems are among the most complex production systems. This is because of the important influence of environmental factors that vary in both time and space and may be highly specific to local site conditions. Also the correlation between inputs (of fertilisers, pesticides, agricultural engineering, etc.) to outputs (of harvested crop, gaseous field emissions, leachate, etc.) is extremely complex and often non-linear in nature – in contrast to most industrial production systems. In LCAs or CFs in the agro-sector, classical data collection or enquiry is not possible and the creation of mean values is complicated and may have limited meaning/applicability.

Customer Requirements

Food companies are increasingly demanding datasets that allow them to distinguish between:

- growing locations worldwide (country/site specific) and explain why differences occur
- agricultural production systems (annual and perennial crops, organic vs. conventional etc)
- production routes from a technical point of view
- product- and packaging designs
- different environmental impacts e.g. account for deforestation and carbon sequestration

The actual data requests that the food industry direct to PE are extremely varied. While some companies require datasets for the raw material at field edge, others require general processing/handling data, while most clients need datasets on final products (Tab. 1).

Food companies often buy their raw materials or products from different locations all over the world e.g. cashew nuts grown in Brazil, India and Africa (those peeled in India due to cost reasons), which makes data collection time-consuming and complicated due to local and very specific site conditions.

Tab. 1: Examples for data requests in the food industry

Datasets on:	**raw material**	**Processing**	**handling**	**final products**
Examples	peanuts cashew nuts potatoes oats milk coffee wheat etc.	juicing concentration spray drying freeze drying grinding milling pressing peeling roasting etc.	cooling freezing transport storage etc.	cheese chocolate drops olive oil sugar apple juice shortening flour etc.

At the same time the customers expect the results:

- to be delivered quickly
- to be highly reliable
- to be easily understandable and marketable – companies want results that can be easily communicated to stakeholders and also want to be able to balance the environmental issues with other areas of concern such as social equality (fair trade, sharing benefits, not exploiting farmers in poor countries, etc.) and affordability (relating to subsidies, competition for land, input costs of fuel and fertilisers, etc.)
- to be delivered at a reasonable cost.

Challenges for LCA/Carbon Footprint Practitioners

Meeting the demands mentioned above is itself challenging but is often further complicated as:

- clients are often not able to provide any data (especially on the agricultural process) from their suppliers

- reliable databases for the food industry are missing, only few basic data sets are already available
- data collection is difficult and expensive
- The concept of an "average" dataset is difficult to define for most agricultural products. The environmental impacts of a particular crop can vary enormously depending upon farming practices (e.g. intensive, vs. extensive vs. organic), the effects of different soil types, indigenous pests, crop rotations and external factors such as annual climatic variation.

Furthermore, clear guidance is lacking from a methodological point of view. While LCA methodology is well defined in the ISO standards [ISO 14040, 2006; ISO 14044, 2006;], for CFs a standard method is not yet agreed on for the assessment of the lifecycle greenhouse gas emissions.

Currently various national and international initiatives have been established, aiming at harmonised calculations and communication rules of Product Carbon Footprints. The first initiative – a single standard method for the assessment of the lifecycle greenhouse gas emissions of goods and services [BSI PAS 2050 (2008)] – was established in the UK in 2007 by the Carbon Trust. The PAS 2050 document defines how life cycle GHG emissions of a product should be measured. The PAS 2050 is a stand-alone open standard being developed in partnership between the Carbon Trust, the UK government Department for Environment, Farming and Rural Affairs (Defra) and BSI British Standards. The PAS 2050 will be launched on the 29th of October.

Similar initiatives are currently ongoing in Germany, France, Japan, Korea and the US [Kim *et al.* (2008)]. On an international level, the World Business Council of Sustainable Development and the World Resource Institute have just launched the GHG Protocol Product and Supply Chain Initiative. Based on this initiative two new calculation and communication guidelines should available by May 2010. Furthermore, a new ISO standard on Product Carbon Footprinting has been announced.

The GaBi Agrarian LCA Model

An extensive, non-linear, complex computing model for plant production has been developed using the GaBi 4 software tool. This allows the user to effectively meet many of the client needs.

The model for all agricultural cultivation systems implemented in GaBi 4 consistently determines the emissions of NO_3 in water and N_2O, NO as well as NH_3 into air for all cultivated species. At the same time emissions from erosion, fire clearing and the reference system as well as the balance of nutrient transfers within crop rotations are consistently realised within this model.

All relevant input materials for the cultivation process itself (commercial fertiliser incl. lime, organic fertiliser, pesticides, seeds including their production and transport) are integrated into the model as cradle-to-gate data sets. Fuel consumption of the field technique is considered, as well as emissions into air out of the engines used. The provision of cultivated products incl. harvests (output) is integrated up to the edge of field or plantation. All relevant processes taking place on the area under cultivation with emissions into air and ground water (lower limit of rooted soil zone) are considered. Heavy metals remaining in soil are considered as emissions to soil and integration of erosive loss of Norg and Corg as well as of nutrients in water are included in the model.

Time reference is a cultivation period from preceding crop to harvest of the respectively considered cultivation fruit / plantation preparation (ground clearing etc.) until optional clearing of the field for further uses. Nutrient transfer between different crops in a rotation (respective plantation pre-usage and further usage) is considered and works on the principle of the delivery of a nutrient usage potential.

The cut-off criteria used are in total <1% of the environmental relevance according to comparative calculations or "expert judgement".

The GaBi 4 agrarian plant model:

1. Can be used to model all types of crop anywhere in the world (different locations and environments)
   - Variations in rainfall and temperature are accounted for in the model
   - Accounts for arable and plantation crops
2. Can model different farming systems; it accounts for
   - Chemical fertiliser and manure use
   - Use of agrochemicals (pesticides, herbicides, fungicides, etc.)
   - Mechanical operations (ploughing, seeding, harvesting, etc.)
3. Covers a range of environmental issues
   - Considers land use changes (deforestation)
   - Carbon sequestration (the carbon balance is properly assessed)
   - Accounts for emissions from erosion, fire clearing and background emissions (soil emissions that would occur whether a crop was planted or not)
   - Considers the balance of nutrient transfers within crop rotations and the use of cover crops Covers impacts such as eutrophication, which play a major role in agricultural production systems
   - Accounts for time dependent features such as rainfall and fertiliser application during parts of the plant growth cycle (Fig. 1).

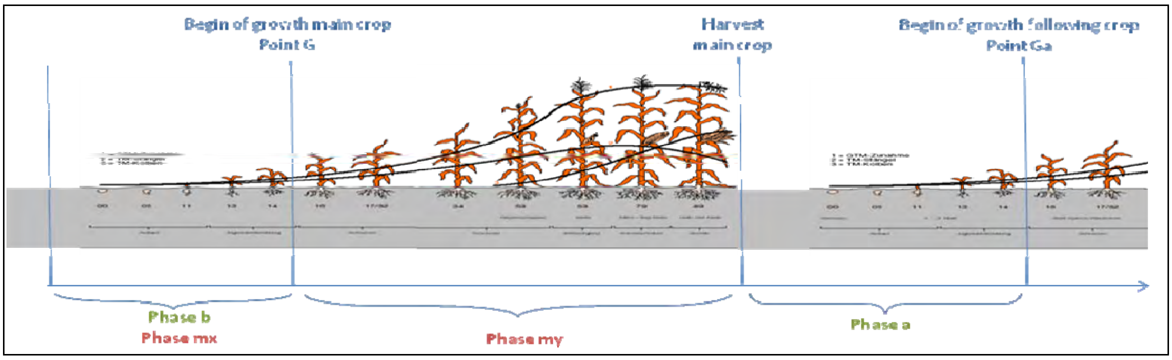

Fig. 1: The time factor being a critical point in plant production

The modelling of CO_2 uptake, the renewable energy storage in biomass and the modelling of nitrogen are key points in the modelling of agricultural products. The following paragraphs will focus on how these issues are considered in the GaBi model.

Modelling of CO_2 uptake: The product-bound CO_2 is directly accounted for being (in the inventory) 100% on the input-side as "Carbon dioxide [Renewable material resources]" identical with GWP factors like "Carbon dioxide [Inorganic emissions in air]". The inventory quantity is calculated as carbon content * 44 / 12. This means that the CO_2 integration into the biomass is NOT included into possible allocations or credits of harvests or further processing, but counted as "feedstock CO_2".The CO_2 quantities emitted during the further life cycle (e. g. at combustion of waste automotive parts made out of renewables) have to be accounted for as CO_2 emissions in air, in the same way as when burning non-renewable materials. The "CO_2 neutrality of the carbon included in products made out of renewables" results automatically there from. Also other C-forms (e.g. CH_4 and CO emissions) are to account for as corresponding emissions in air (e.g. release of methane from waste dumps or by incomplete combustion).

Modelling of renewable energy storage: The storage of renewable energy (finally sunlight) in agro-products is calculated as lower calorific value and accounted for as "Primary energy by sun [Renewable energy resources]" with the base quantity "Energy reg. (lower calorific value)" input-bound - by analogy with the CO_2 embodied in the renewable products. Thereby the energy embodied in the product is accounted for irrespective of the products' water content.

Modelling the nitrogen cycle: This is the most complex aspect of the model and affects a number of key emissions having environmental relevance in most LCA studies including NO_3- in water and N_2O, NO and NH_3 into air (Fig. 2) The figure shows systematically the most important N-flows; the arrows' or depot-boxes' width hereby corresponds to the approximate quantity of N per year - illustrated by an intensive cultivation system of a cereal crop. Attention should be paid to the fact that these values may be extremely different among crops and cultivation systems and that the emissions, which are finally relevant for the inventory results from the difference between N-input and N-output.

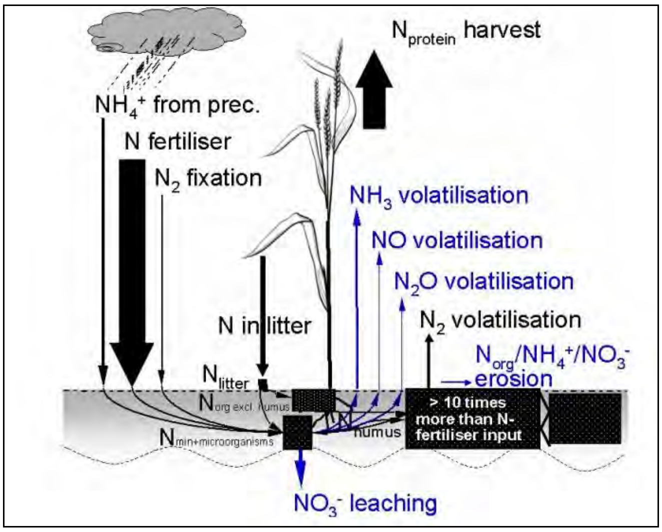

Fig. 2: Key aspects of the nitrogen cycle

The N-modelling underlies different assumptions. It is understood that Ninput = Noutput i.e. the balance is nil for the examined crop. If there is, mathematically, a net N-reduction in the soil (due to less fertiliser input and additional supply from soil) or an N-accumulation (as Nmin or organic material in soil), this will be balanced by additional/reduced external fertiliser demand. Thereby, the amount of N being fixed in humus in the long run is assumed as constant. This excludes (only apparently) very environmentally beneficial cultivation systems without fertiliser application which only work at cost of the nutrient pool in soil and which they deplete thereby reducing the growth potential of the site. In the last instance, the consumed net nutrient removal must be balanced, which is realised here by the above-mentioned integration of an external fertiliser requirement. At the same time, an abundant quantity of nutrient surplus remaining in soil, which is available for the follow-up crops - according to the actual utilisation potential - is credited to the examined crop.

The Agrarian Model (and the associated databases) within GaBi 4 allows the user to map the raw material production of any plants produced worldwide for direct use in food products or used as fodder in animal production for final products (such as milk, cheese, meat etc.). Especially in animal derived products where the refinement factor from plants (fodder) to the final product (such as meat), is quite high, the agrarian model takes care of the impacts related to fodder production.

Discussion

The Agrarian model meets many of demands from stakeholders in terms of providing robust data on the environmental impact of food products. However, some difficult challenges remain in some areas -

agricultural systems are complicated, so the model developed to assess them is also complicated and both data- and resource-intensive to use. As such, assessments of agricultural products remain time consuming and expensive.

How can retailers, who may stock thousands of food items but who do not manufacture these themselves, effectively assess carbon footprints and other environmental impacts of this range of products? There is a need for a streamlined data collection process or central resource where the data requirements of the model can be easily accessed.

Conclusion

There is increasing demand from stakeholders across the supply chain for a better understanding of environmental impacts of agriculturally-derived products. New tools such as PE's Agrarian model are being developed to provide this information.

Challenges remain in terms of enabling the rapid and cost effective environmental assessment of agricultural products.

From a methodological point some uncertainty will remain in the upcoming years due to the range of competing approaches and standards being adopted in different countries. However, what all these new initiatives have in common is that they refer to the well established ISO standards [ISO 14040, 2006; ISO 14044, 2006;] as a point of departure and that they aim to be compliant with the ISO 14040/44. The main focus is to give further specifications with respect to carbon specific issues such as carbon sequestration and handling of biogenic carbon.

Thus, companies can start the process of carbon footprint implementation today, by following the ISO 14044 standard.

Methodological uncertainties can easily be addressed if appropriate software tools such as GaBi 4 are employed in the implementation process. For instance, various allocation scenarios are straightforward to calculate using criteria such as energy, mass and economic criteria.

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Multi-Criteria Analysis on Countermeasures against Livestock Manure Excess Supply Problem in Maebashi City, Japan

T. Iwata and S. Shimada
Department of Environment Systems, Graduate School of Frontier Sciences, The University of Tokyo, 5-1-5 Kashiwanoha, Kashiwa, Chiba, Japan.

E-mail: t-iwata@k.u-tokyo.ac.jp

Keywords: Livestock manure, Policy, Material Flow Analysis, Multi-Criteria Analysis, Simulation model

Abstract

Japanese government and local authorities propose some countermeasures with subsidy to cope with the livestock manure excess-supply problem. However, the implementation cost is so high that an effective allocation of governmental subsidy is needed.

In this study a budget allocation model for reasonable policy planning on manure excess-supply problem was developed based on Multi Criteria Analysis and Life Cycle Impact Assessment. As a case study, the expected effects of optimised budget allocation were compared with the effects of actual budget allocation of Maebashi City in 2007.

The results based on numerical model simulation indicated that the Maebashi plan (2007) gave the priority to improvement of local environment. The results also indicated that the combination of feed production project, methane fermentation and livestock reduction was the most effective to increase social & environmental benefits that were important benefits for residents in Maebashi area. However, more discussion about the introduction of Policy 1 ("Livestock reduction") should be done with consensus building between government and farmers because Policy 1 has not accepted and introduced in Japan yet.

1. Introduction

Japanese livestock farmers have been expanding the farm scale with an imported feed to reduce the production cost and manpower. As a result, a large amount of livestock manure is emitted in each livestock farm without any utilization because of the lack of their own agricultural field. Livestock manure was utilized effectively as a good fertilizer by most of field farmers in the past. However, the manure demand has been decreased recently because of the spread of an imported chemical fertilizer, which is cheaper and easier to handle than manure. Therefore, livestock manure is in a state of excess-supply (Fig. 1). This problem is very serious in livestock congested area, such as Miyazaki, Kagoshima and Gunma prefecture in Japan.

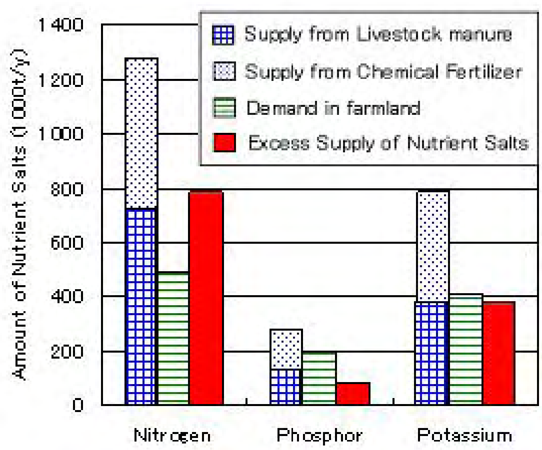

Fig. 1 Balance of Nutrient Salts in Japan

To cope with this problem, Japanese government and local authorities propose some countermeasures with subsidy, such as the promotion of domestic feed production, construction of manure disposal plant, methane fermentation plant, and so on. Those countermeasures are expected to alleviate the manure excess-supply problem, however, the implementation cost is so high that an effective allocation of governmental subsidy is needed.

For the proposal of a reasonable policy planning with an effective government budget allocation, Multi-Criteria-Analysis (MCA) has attracted attention as a useful method. MCA appeared in the 1960s as a decision-making tool. The method is designed to help decision-makers to integrate the different options, reflecting the opinions of the actors concerned, into a prospective framework. As a useful tool to make a comparative assessment of alternative projects or heterogeneous measures, MCA has been applied for an evaluation of public works projects. The importance of MCA application to agriculture has been mentioned[1][2], however, the application based on material flow model has not seen yet in Japan.

2. Purpose of the study

The purpose of this study is to propose the methodology for reasonable policy planning on manure excess-supply problem based on MCA. The main steps involved in MCA can be broken down into several phases [3].

- Phase 1. Identifying of the Projects or Actions to be judged
- Phase 2. Identifying the Alternatives (Countermeasures in this study)
- Phase 3. Identifying the Criteria of Evaluation Terms
- Phase 4. Scoring the Evaluation Terms in Relation to the Criteria
- Phase 5. Weighting the Scores According to the Weights Assigned to the Criteria
- Phase 6. Evaluating the Alternatives with a single synthetic unit calculated by scores and weights

In the evaluation of the environmental effects of countermeasures, Life Cycle Impact Assessment (LCIA) is useful tool. LCIA has also similar structure to MCA. Many kinds of methods for Life Cycle Impact Assessment are proposed such as LIME, Eco-Indicator and EPS. Those methods can show the result of assessment as a single synthetic unit. LCIA was originally developed to assess the environmental aspects and potential impacts of a product through a product's life (cradle to grave), however, the target of LCIA has been expanding from product to system.

The difference between LCIA and MCA is the contents of “Impact category”. The “Impact category” in LCIA includes only environmental factors because the “Characterization process” can’t be applied to social factors. “Characterization process” is basically determined by scientific knowledge, therefore it might be hard to “Characterize” a social benefit. (Recently Social LCA has been discussed to evaluate not only environmental and economic aspects but social aspect[4].) On the other hand, MCA is more comprehensive method than LCIA. The scoring process in Phase 4 of MCA includes the scoring by both quantitative and qualitative scale, therefore MCA can includes the evaluation of social impacts. LCIA can be applied to the part of scoring (especially environmental impact), therefore we combined LCIA to MCA to evaluate the environmental & social impacts caused by countermeasures against livestock manure excess-supply problem[5].

In this study we set the amount of allocated budget as variables and maximized the single synthetic unit in Phase 6 as an objective function to propose an optimal budget allocation. As a case study, an effective government budget allocation for livestock manure excess-supply problem in Maebashi, Gunma prefecture, Japan was proposed with the developed optimization model. The expected effects of optimised budget allocation were compared with the effects of actual budget allocation of Maebashi in 2007.

3. Outline of Budget Allocation Model

3.1 Candidates of countermeasures

There are five feasible policies to cope with the excess livestock manure (Fig. 2).

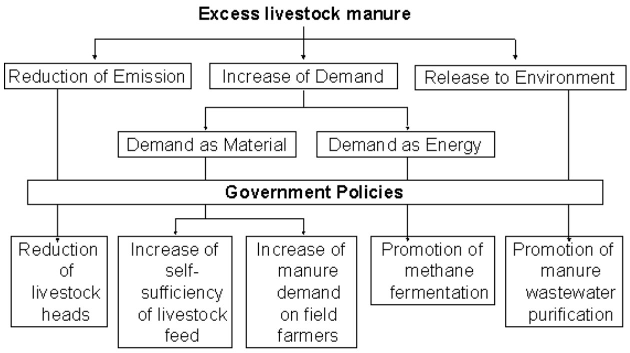

Fig. 2 Candidates of countermeasures

Policy1: Reduction of livestock heads

Policy1 aimed to reduce manure emission. EU countries have already implemented the control of livestock heads to avoid manure excess emission. Regulation of livestock heads control could be implemented in the future in Japan.

Policy2: Increase of domestic feed production

Policy2 aimed to increase manure demand as fertilizer resource. Japanese government plans to help feed production in abandoned cultivated land instead of feed import.

Policy3: Increase of manure demand on field farmers

Policy3 aimed to increase manure demand as fertilizer resource. Japanese government plans to help compost distribution between livestock farms and field farms instead of chemical fertilizer import.

Policy4: Promotion of methane fermentation

Policy4 aimed to increase manure demand as energy resource. Methane fermentation plant produces bio-methane gas as domestic renewable energy.

Policy5: Promotion of wastewater purification

Policy5 aimed to dispose manure without any utilization based on environmental standard for water quality. Excess manure is now left on the field. Construction of manure & wastewater purification plant can stop the environmental pollution from excess manure which is now left on the field.

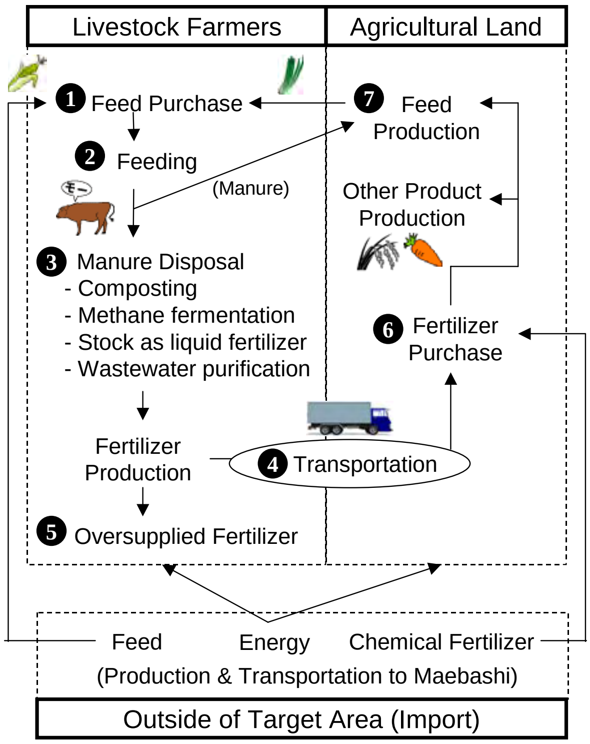

Fig. 3 System boundary of "Sub-Model for Agricultural Material Flow Analysis"

3.2 Sub-Model for Agricultural Material Flow Analysis

This model consists livestock manure supply-demand balance, livestock feed supply-demand balance and fertilizer supply-demand balance (Fig. 3). To simulate the fertilizer transportation routing between livestock farm and agricultural field, all databases was based on Geological Information System (GIS) in this model. Maebashi city was divided into 42 area based on GIS, and the transportation of manure compost between areas was simulated. Main equations were showed below. Fertilizer effectiveness ratio was already included in the amount of fertilizer demand.

Main equations of material flow[5]:

$$\sum_{t=1}^{4}\sum_{a=1}^{42}\mathrm{Feedsupply}(t,a,f)\,(t/y)=\sum_{a=1}^{42}\mathrm{Domesticfeedproduction}(a,f)\,(t/y)+\mathrm{Importedfeed}(f)\,(t/y)$$

Manure emission(t,a) (t/y) = {Initial number of livestock(t,a) - Reduction number of livestock(t,a)} (head) ×manure emission unit(t) (t/y/head) = Composting(t,a) + Methane fermentation(t,a)

Urine emission(t,a) (t/y) = {Initial number of livestock(t,a) - Reduction number of livestock(t,a)} (head) ×Urine emission unit(t) (t/y/head) = Stock(t,a) + Methane fermentation(t,a) + Wastewater purification(t,a)

Compost production(t,a) (t/y) = Composting(t,a) ×Compost production ratio (t/manure-t)

$$= \sum_{a'=1}^{42} \text{Compost transportation amount}(t,a,a') + \text{Non-utilized compost}(t,a)$$

$$\text{Compost demand}(t,a',ns)\,(t/y) \geq \sum_{a=1}^{42} \text{Compost transportation amount}(t,a,a') \times \text{Compost nutrition}(t,ns)\,(t/compost\text{-}t) + \text{Chemical fertilizer}(a',ns)\,(t/y)$$

$$\text{Fast-acting fertilizer demand}(a,nf)\,(t/y) \geq \sum_{t=1}^{4} \{\text{Stock}(t,a) \times \text{Urine component}(t,nf)\,(t/head) + \text{Methane fermentation}(t,a) \times \text{Digested sludge component}(t,nf)\,(t/sludge\text{-}t)\} + \text{Chemical fertilizer}(a,nf)\,(t/y)$$

Main equations of cost calculation:

Cost for “Policy 1” (JPY/y) =

$$\sum_{t=1}^{4}\sum_{a=1}^{42} \text{Reduction number of livestock}(t,a) \times \text{Compensation cost of livestock}(t)\ (JPY/head/y)$$

$$\text{Cost for “Policy 2” (JPY/y)} = \sum_{a=1}^{42} \text{Domestic feed production}(a,f) \times \text{feed production cost}\,(JPY/t)$$

$$\text{Cost for “Policy 3” (JPY/y)} = \sum_{t=1}^{4}\sum_{a=1}^{42}\sum_{a'=1}^{42} \text{Compost transportation amount}(t,a,a') \times \text{Distance between a and a'}(a,a')\,(km) \times \text{Transportation cost}\,(JPY/t/km)$$

$$\text{Cost for “Policy 4” (JPY/y)} = \sum_{t=1}^{4}\sum_{a=1}^{42} \text{Methane fermentation}(t,a) \times \text{Plant construction cost}\,(JPY/t/y)$$

$$\text{Cost for “Policy 5” (JPY/y)} = \sum_{t=1}^{4}\sum_{a=1}^{42} \text{Wastewater purification}(t,a) \times \text{Plant construction cost}\,(JPY/t/y)$$

t = Livestock type (Milk cow, Beef, Pig, Chicken) (t = 1-4)

a = Area number (j = 1-42)

f = Component index of livestock feed (Total Digestible Nutrients, Neutral Detergent Fiber)

m = Manure disposal method (Composting, Methane fermentation)

u = Urine disposal method (Stock, Methane fermentation, Wastewater purification)

ns = Fertilizer Nutrition (Slow-acting N,P,K)

nf = Fertilizer Nutrition (Fast-acting N,P,K)

a’ = Area number of compost demand (j=j’= 1-42)

3.3 Sub-Model for Characterization of Environmental & Social Benefits

The categories of environmental & social impacts were determined by Panel method[5]. For the scoring of each category, "Characterization Process" in LCIA was applied to the index of "Water Pollution", "Global Warming" and "Acidification"[6] (Fig. 4). For the calculation of each index, "Inventory Analysis" in LCA was done based on "Sub-Model for Agricultural Material Flow Analysis"(Tab. 1). An improvement of the logic in index setting is the future task of this study.

Tab. 1: Boundary of inventory analysis (Process 1-7 were shown in Fig. 3)

Impact Category		Inventory	Process						
			①	②	③	④	⑤	⑥	⑦
Local	Water Pollution	NO_3-N					○		
		P					○		
	Soil Pollution	Cd					○	○	
	Odor Problem	NH_3		○	○				
Social	Food Self-Sufficiency Ratio	Meat/ Feed Production	○	○					
	Energy Self-Sufficiency Ratio	Energy Consumption / Production	○	○		○		○	○
Global	Acidification	NH_3		○	○				
		NOx	○			○			○
		SOx	○			○			○
	Global Warming	CO_2	○	○		○		○	○
		CH_4		○	○		○		
		N_2O			○		○		
	Exhaustible Resource Protection	P	○					○	

Local environmental Impact

Index of Water Pollution (PO_4-eq.)
= 0.42× Nitrate Leaching (NO_3-N) + 3.06× Excess Phosphorous (P)

Index of Soil Pollution (Cadmium) = Input Cd Amount (Cd)

Index of Air Pollution (NH_3) = Ammonia Emission (NH_3)

Social Impact

Index of Feed Self Sufficiency
= (-1)× (Livestock Head / Initial Livestock Head)× (Feed-TDN Production / Feed-TDN Demand)

Index of Energy Self Sufficiency = Energy Demand – Energy Production

Global environmental Impact

Index of Global Warming
= 1× Carbon Dioxide (CO_2) emission + 23× Methane (CH_4) emission
+ 296× Nitrous-Oxide (N_2O)

Index of Acidification
= 1× Sulfur Dioxide (SO_2) emission + 1.88× Ammmonia (NH_3) emission
+ 296× Nitrogen Oxide (NO_x)

Index of Exhaustible Resource = Phosphorous Consumption (P)

Fig. 4: Definition of impact category indices (The index of "Food self-sufficiency" was multiplied by (-1) to show the benefit as a positive value as well as other indices)

3.4 Sub-Model for MCA

As a single synthetic unit in Phase 6, "Evaluation Value" was set in this model. Each scores calculated by "sub-model for characterization of environmental & social benefits" were reflected to below "Indicator". The definition of "Indicator" was based on Goals-Achievement Method[3]. The result of maximization showed the optimal budget allocation which could increase the benefits of highly weighted terms. In this study the total amount of budget was set about 1.6 million US$ (this is actual budget for livestock excess manure problem in Maebashi (2007)) as a constraint of this model. In this study the subsidy ratio of "Policy 1" was set as 100(%) and others were set as 50(%).

Objective Function : Maximize (Evaluation Value)

$$\text{Evaluation Value} = \sum_{i=1}^{8} Weight_i \times Indicator_i$$

$Weight_i$ = Weight of each "Impact category"

$$Indicator_i = \frac{V_i - (V_i)_{initial}}{(V_i)_{\max} - (V_i)_{initial}}$$

V_i = Value of each "Impact category" index

$(V_i)_{max}$ = Maximum Value of each "Impact category" index under budget constraint

$(V_i)_{initial}$ = Initial Value of each "Impact category" index

i = "Impact category" : i = 1 ~ 8

Constraint : Budget for livestock manure problem in study area $\geq \sum_{j=1}^{7} BudgetAllocation_j$

Budget Allocation $_j$ = Cost for each policies (j) × Subsidy ratio (j)

j = Policy number (j = 1-5)

4. Case study in Maebashi City

4.1 General Information of Maebashi City[7]

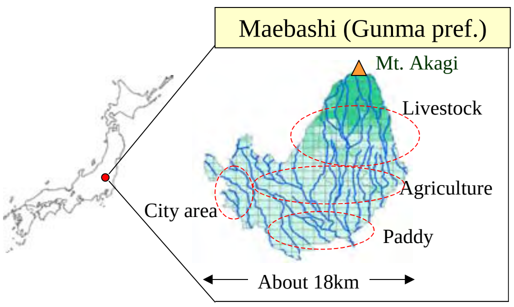

Fig.5. Location of study area; Maebashi city

The area of Maebashi is 241.22(km^2) and the population is 31,967 people. Maebashi has a typical inland climate and the average of temperature ranges from about 14 ˚C ~ 15 ˚C. Fig. 5 shows the location of Maebashi City in Japan.

There are 9 thousands of dairy cow, 15 thousands of beef, 148 thousands of pig, and 654 thousands of chicken. The total livestock manure emission is about 2,500(ton/day). There are about 8290(ha) of agricultural field and the area for feed production is 1400(ha). In Maebashi the livestock manure excess problem is a serious environmental and social problem.

4.2 Weighting of "Impact category" by AHP based on the results of questionnaire survey

The weight of each "Impact category" was calculated by Analytic Hierarchy Process (AHP) based on questionnaire survey. The provided information in the questionnaire survey has a strong influence on the result because the judgment of the priority between the "Impact category" is not easy generally. In this study we provided little information about each "Impact category" to get the result which was strongly influenced by their current environment such as TV, newspaper, education and so on. The questionnaire survey was done two times. The weight of each "Impact category" was calculated by AHP[8] (Fig. 6). The result of weighting was shown in Fig. 7. The weight of "Water Pollution" and "Feed Self Sufficiency" were relatively high in both of survey because Maebashi area is located near a big river (Tone River) and the main industry in Maebashi is agriculture.

Survey 1.

Surveyed:

1,000 residents in Maebashi

(Simple Random Sampling from NTT phonebook)

Survey method :

By mail (No reminder)

Survey Period:

26/01/2007-05/02/2007

Response Rate : 20.8 %

(Valid Response Rate: 20.1%)

Survey 2.

Surveyed:

1,000 residents in Maebashi

(Simple Random Sampling from NTT phonebook)

Survey method :

By mail (No reminder)

Survey Period:

15/11/2007-30/11/2007

Response Rate : 18.3%

(Valid Response Rate: 16.9%)

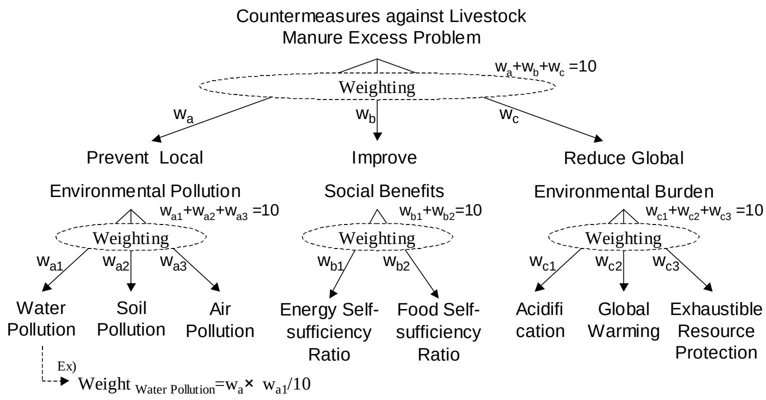

Fig. 6 Outline of weighting in AHP

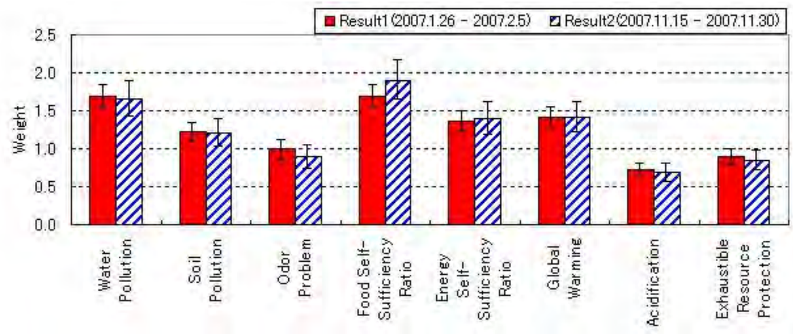

Fig. 7 Result of weighting to "Impact category" based on questionnaire survey

4.3 Indicator setting based on Goals-Achievement Method

$$Indicator_i = \frac{V_i - (V_i)_{initial}}{(V_i)_{\max} - (V_i)_{initial}}$$

$$\begin{cases} V_i = \text{Value of each “Impact category” index} \\ (V_i)_{max} = \text{Maximum Value of each “Impact category” index under budget constraint} \\ (V_i)_{initial} = \text{Initial Value of each “Impact category” index} \\ \text{i = “Impact category” : i = 1} \sim 8 \end{cases}$$

The $(V_i)_{initial}$ showed a current index of each "Impact category". A current agricultural system in Maebashi was input into the developed model as a parameter. Then the model calculates each index of environmental & social impacts in the current system. To get the $(V_i)_{max}$ under budget constraint (1.6 million US$) the objective function was modified to "Maximize "Impact category" index(i)". The difference in the index $(V_i)_{initial}$ and $(V_i)_{max}$ showed the range which could be improved by countermeasure(Tab. 2).

Tab. 2: Result of the index $(V_i)_{initial}$,$(V_i)_{max}$ and the range which could be improved by countermeasure.

Impact Categories	$(Vi)_{initial}$	$(Vi)_{max}$	Range of Improvement	(Unit)
Water Pollution	9,530	5,706	3,824	(kg- PO_4- eq/ day)
Cadmium Pollution	298,194	202,034	96,160	(mg- Cd/ day)
Odor Problem	4,297	3,171	1,125	(kg- NH_3/ day)
Global Warming	565,009	504,670	60,338	(kg- CO_2- eq/ day)
Acidification	13,757	11,624	2,133	(kg- SO_2- eq/ day)
Phosphorous Exhausion	4,592	3,570	1,021	(kg- P/ day)
Food Self-sufficiency Ratio	610	851	241	(no unit)
Energy Self-sufficiency Ratio	312,506	298,310	14,196	(MJ/ day)

4.3 Results and Discussion

As the solver of this model, we used NUOPTver.6. The number of variables was 8002 in this model. With the developed optimization model the effective government budget allocation for livestock manure excess-supply problem in Maebashi was proposed and the expected effects of optimized budget allocation was compared with the effects of actual budget allocation of Maebashi in 2007 (Fig. 8).

The result of the simulation showed that the optimized budget allocation was 58.8% to Policy1; "Reduction of livestock heads", 19.1% to Policy2; "Production of feed", 7.7% to Policy3; "Compost transportation" and 14.4% to Policy4; "Methane fermentation". In this plan, the budget for Policy1 contributed to reduce 2768 pig heads which was about 1.8% of total pig heads and the budget for Policy2 contributed to domestic feed production in 456 (ha) which was about 88% of total abandoned land.

In Maebashi plan (2007) the budget was mainly allocated to Policy 3. On the other hand in optimized plan the amount of allocated to Policy 3 was very small because of the constraint of fertilizer nutrients supply- demand balance. In Maebashi the supplied nutrients from manure was further larger than the demand in agricultural field. In a current system, manure compost was utilized enough so that there was no need to allocate big budget to Policy 3. Policy 5 was not introduced in the optimized plan

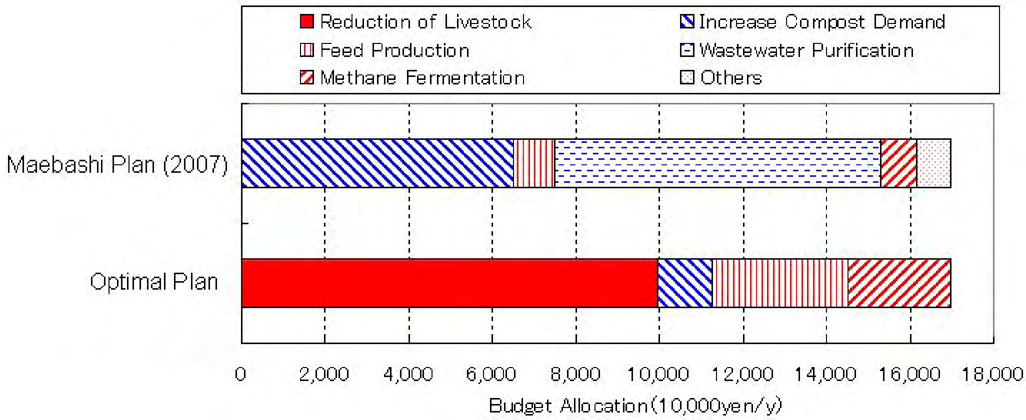

Fig. 8: Result of the budget allocation in Maebashi Plan (2007) and Optimized Plan

because of an environmental problem.

Fig. 9 shows the each index of “Impact category” in Maebashi plan (2007) and the optimized plan. The index of “Global Warming” in Maebashi plan (2007) was seriously worse than current system because of the introduction of wastewater purification plant. However, most of the indexes of local environmental pollution such as “Water Pollution”, “Soil Pollution” and “Oder problem” could be relatively improved in comparison with the optimizes plan. On the other hand, the optimized plan could improve totally including not only the local environmental pollution but the global environmental burden (“Global Warming”, “Acidification” and “Resource Protection”) and social benefits (Self-Sufficiency Ration of Food and Energy).

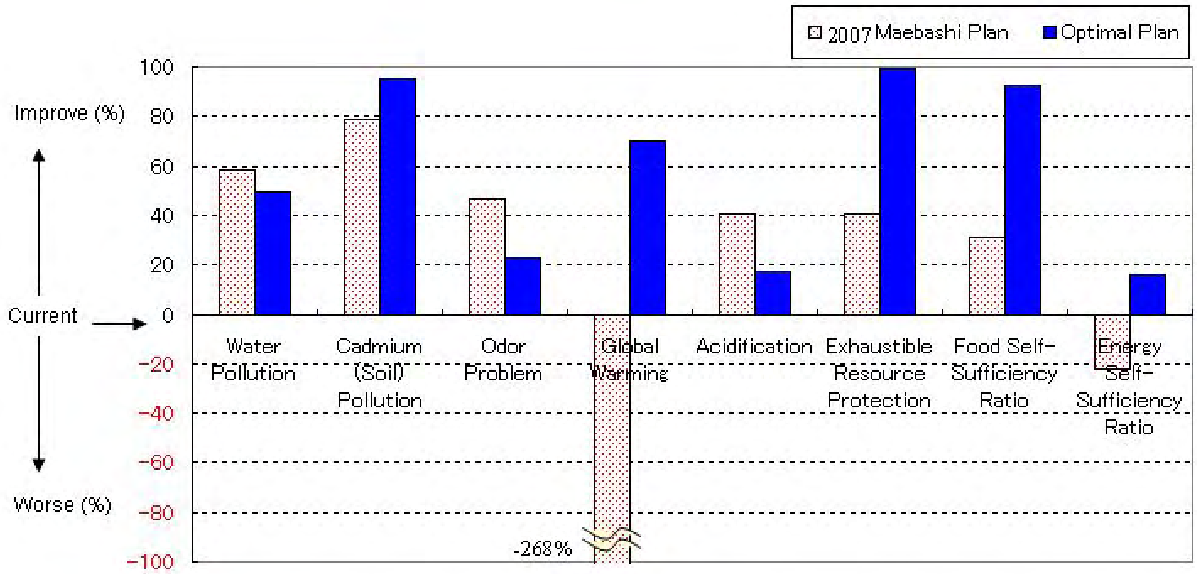

Fig. 9 Result of the each index of “Impact Category” in Maebashi Plan (2007) and Optimized Plan

The result indicated based on numerical model simulation that the Maebashi plan (2007) gave the priority to improvement of local environment. The model suggested the optimized plan including Policy1 (“Reduction of livestock”) which could contribute to improve the environmental & social benefits totally, however, more discussion about the introduction of Policy 1 should be done taking into account of consensus building between government and farmers in the future works because Policy 1 has not accepted and introduced in Japan yet.

5. Conclusion

In this study a budget allocation model for reasonable policy planning on manure excess-supply problem was developed based on Multi Criteria Analysis and Life Cycle Impact Assessment. As a case study, the expected effects of optimised budget allocation were compared with the effects of actual budget allocation of Maebashi City in 2007.

The results also indicated based on numerical model simulation that the Maebashi plan (2007) gave the priority to improvement of local environment. The results also indicated that the combination of feed production project, methane fermentation and livestock reduction (Policy 1) was effective to increase social & environmental benefits which were important benefits for residents in Maebashi area. However, more discussion about the introduction of Policy 1 should be done taking into account of consensus building between government and farmers in the future works because Policy 1 has not accepted and introduced in Japan yet.

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Estimating the carbon footprint of raw milk at the farm gate: methodological review and recommendations

C. Basset-Mens [1, 2]
1: AgResearch Limited, Ruakura Research Centre, East street, Private Bag 3123, Hamilton, New Zealand
2: Cemagref, UMR ITAP, 361 rue Jean-François Breton, F-34196 Montpellier, France. Tel.: +33 4 67 04 63 86. E-mail: claudine.basset-mens@cemagref.fr

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Abstract

The demand for reliable LCA data has become very compelling in the food sector. This paper presents a methodological review of cradle-to-farm-gate LCA studies for the GWP of raw milk as a typical case study. All steps of the LCA methodology have been included. Despite a quite consistent range of results across studies (0.85 – 1.4 kg CO_2-eq/kg Energy Corrected Milk), this analysis concludes that comparing independent LCA studies is questionable. Beyond the use of different key assumptions (FU, allocation rules, GWPi) to be harmonised, a lack of transparency and consistency exists in the description of farming systems and inventory methods and data. Firstly, the definitions of key parameters and the reporting of LCA studies need to be harmonised. Secondly, more in depth collaborative work could focus on harmonising methods for the inventory analysis. This could be the mission of international working groups specialised per product category. A far more challenging task would be to succeed in obtaining national research programmes on key knowledge gaps at a global level such as N_2O emissions from soils. Ambiguously at this stage, it is not possible to conclude whether results are consistent and/or different for real reasons or for method reasons.

Introduction

The food industry is receiving a growing pressure from consumers, retailers and governments to produce environmentally-friendly products over their whole life cycle. Among all environmental impacts, the Global Warming Potential (GWP) in particular, also called "carbon footprint", is currently given a stronger emphasis with specific norms being developed such as the UK-based Carbon Trust methodology. In this context, New Zealand dairy companies for instance, exporting dairy products worldwide are looking at benchmarking the carbon footprint of their products along their whole life cycle using methods able to stand international scrutiny. Dairy products have been the most studied of all food products using the life cycle assessment (LCA) methodology. However, in order to produce reliable comparisons of the carbon footprint of dairy products across countries, the consistency and harmonisation of all assumptions, models and references used across all studies compared need to be checked, especially for the cradle-to-farm-gate stages representing the major contributor to the GWP of dairy products. De Boer (2003) already highlighted how comparing LCA studies can be an uncertain exercise. The purpose of this review is to answer the question: can we use independent (attributional for the moment) LCA studies across countries with similar goal at their national/regional level to conclude on the actual differences in terms of environmental performance between typical production systems from different regions of the world?

LCA results from independent studies (Tab. 1) can differ for several reasons:

1) First, the studied systems are different
2) The assumptions made are different, some of which could possibly be harmonised across studies
3) The scientific knowledge regarding the direct emissions from a given system differs from one country to the other due to their different research priorities and achievements.

4) Some calculations errors occurred and might be very complicated to identify once the study has been completed and reported
5) For regional impacts (so not for GWP), the local/regional ecosystems have different sensitivities if this dimension has been integrated in the assessment so far, for instance through models calibrated in specific soil and climate conditions or integrating specific features of the country.

Based on a selection of cradle-to-farm-gate LCA studies, this review focuses on methods and data used to estimate the global warming potential (GWP) for raw milk production in different countries. The objectives are:

- To identify key discrepancies between methods and assumptions used
- To conclude on the possibility to reliably compare independent LCA studies
- To make some recommendations on possible harmonisation and improvement options

Methodological review

Over the last ten years, scientific production on life cycle assessment applied to milk and dairy products has been significant relative to LCA studies for other food products. For this methodological review, papers showing a similar goal but a different geographical scope were selected. The criteria of selection for papers were as follows:

- Exhaustive and well reported LCA or carbon footprint study
- Studied system from cradle-to-farm-gate
- Designed to be representative of raw milk production for a given country

This selection process resulted in a short-list of seven papers. Papers dealing with more specific methodological aspects, too specific dairy system scenarios or with systems going beyond the farm gate (such as Eide, 2002, Berlin, 2002, and Hospido *et al*., 2003) were excluded.

Goal and scope

In all selected papers, the goal and scope of the study is clearly defined. Within a given country the goal is to gain knowledge about the environmental impacts of raw milk production in its most typical conditions of production at the national level and often to explore the effect of different rules of production, practice and management options (Tab. 1). Results for the different systems can be either presented separately or presented as a weighted average according to their ratio in the country (Williams *et al*., 2006). In most studies, the functional unit is one (or 1000) kg of energy corrected milk (ECM) also called fat and protein corrected milk, except Haas *et al*. (2001) where quality is not specified and Williams *et al*. (2006) using a functional unit of 10 m^3 of fat-corrected milk. Allocation between milk and other co-products on dairy farms varies between economic allocation (Cederberg and Flysjö, 2004; Williams *et al*., 2006, Thomassen *et al*., 2008), biological causality (fodder requirement) (Cederberg & Mattsson, 2000; Basset-Mens *et al*., 2008), and no allocation at all (Haas *et al*., 2001) (Tab. 1). Casey and Holden (2005) make a sensitivity analysis to the allocation rules by assuming none, mass or economic allocations and use economic allocation for their final result.

Generally speaking the scope of the selected studies is consistent in terms of period (2000-2005) and technology. Conventional production systems are always studied, sometimes in comparison with organic and/or other more “extensive” systems (Tab. 1). Most studies differ in terms of geographical location, except for the two Swedish studies (Cederberg and Mattsson, 2000; Cederberg and Flysjö, 2004). The system boundaries across all studies is quite consistent with minor inputs such as capital or pesticides being either included or excluded while key inputs such as on-farm processes, feed, fertilizers and energy sources being always included in the analysis.

Inventory analysis

Dairy farm systems

The design of the studied farm systems and their technical data are based on a range of approaches from the survey of a sample of farms (studies have used between 1 and 11 farms per production type) to the use of national statistics and database (Tab. 2). This range of approaches would need to be analysed for its potential effect on the results for a typical scenario at the national scale. Furthermore, it proved difficult to find detailed and consistent information about farm characteristics across selected studies. Certain data were deduced from other data presented or even asked directly to the authors.

The data most consistently presented across studies is milk production per cow as illustrated in Tab. 2 for all conventional systems. Different systems are labelled as “intensive” or “extensive” within a given country which is not sufficient to rank the systems across all studies in terms of intensification degree and typical practice. Key missing technical parameters across studies are most often: the size of the farm, stocking rate, replacement rate, and probably more importantly the total dry matter intake and its spreading between different feed types. Certain information is given in a national unit such as the stocking rate (see Dutch and German Livestock Units) or also the “DVE” describing intestine digestible protein content of feed with a quoted reference in Thomassen *et al*. (2008).

Various terminologies are also used across studies (especially for feed types) complicating the work of summary and comparison between all studied systems. Since qualitative or inconsistent information is given to describe the studied systems in selected papers, the possibility of interpretation of the results in relation to the different systems representative of each country is reduced. The analysis of farm characteristics represents the first way to check the consistency of LCA results. Key parameters to characterise dairy farms and their intensification degree should be more consistently defined and reported.

Environmental inventory of greenhouse gas emissions

In this part, we analyse the information given on the references and methods used for the inventory of each emission as well as the presentation of the inventory data itself.

Methane and nitrous oxide components

In four of the papers detailed and explicit description of the methods used for their greenhouse gas inventory is given (Cederberg and Flysjö, 2004; Casey and Holden, 2005; Thomassen *et al*., 2008; Basset-Mens *et al*., 2008). In the three other studies, general references such as IPCC (1997) are used and quoted without much detail thus making it impossible to know which interpretation of the IPCC (1997) guidelines has been made and whether or not all components (such as indirect N_2O emissions or CH_4 emissions due to manure management) have been accounted for. A global list of references is provided in Haas *et al*. (2001) with no specification of which aspects of the inventory they cover (Tab. 3).

Only Williams *et al*. (2006) strictly applied the IPCC international method for greenhouse gas inventory. Basset-Mens *et al*. (2008) applied the method of their national IPCC greenhouse gas inventory which relies on specific data and emission factors for New Zealand while in the other selected studies a mix of IPCC references and more specific references on emission factors for dairy farm systems in their countries is used (Tab. 3). Therefore, even when using the IPCC method for one country, a discrepancy can still exist due to the different levels of knowledge integrated in each national inventory and agreed by IPCC. This discrepancy exists for instance for the N_2O emission factor for excreta applied during grazing between (probably) all studies and the New Zealand inventory used in Basset-Mens *et al*. (2008). The New Zealand emission factor is 1% instead of 2% for the corresponding default emission factor.

Tab. 1: Studied system, system boundaries, studied function and milk quality, allocation rules and GWP results (as kg CO_2-eq/kg ECM or milk as defined in each study) across selected LCA studies for raw milk.

	Studied system	System boundaries	FU	Allocation rules	GWP *(as a % of NZ result)*
Cederberg & Mattsson (2000)	Representative organic and conventional Swedish production	Cradle-to-farm-gate (buildings, machinery and medicines excluded)	1000 kg ECM*	Biological causality (fodder requirement): 85% milk/15% meat; mass allocation for farmland area and economic allocation for feed ingredients	1.100 *(123%)* 0.950 *(111%)*
Cederberg & Flysjö (2004)	Representative conventional high, conventional medium and organic milk production in Sweden	Cradle-to-farm-gate (buildings, machinery and medicines excluded)	1000 kg ECM*	Economic allocation at all levels: 90% milk/10% meat	0.896 *(105%)* 1.037 *(121%)* 0.938 *(110%)*
Haas *et al.*, (2001)	Representative German intensive, organic and extensive milk production	Cradle-to-farm-gate (buildings, machinery excluded)	1 kg milk (quality unspecified), ha of farmed grassland, whole farm#	None (meat production considered not significant enough but still estimated to be about 10%)	1.3 *(152%)* 1 *(117%)* 1.3 *(152%)*
Casey and Holden (2005)	Average Irish milk production	Cradle-to-farm-gate	1 kg ECM*	Economic	1.3 *(152%)*
Williams *et al.*, (2006)	Representative English and Wales milk production, weighted average milk between conventional/organic/alternative at 3 yield levels for each type*	Cradle-to-farm-gate	$10m^3$ milk (apparently as fat-corrected milk)	Economic for milk and feed ingredients; maintenance cost of cows avoided when dairy bred calves enter beef sector; 50% of available N in slurry used to save fertiliser	1.03§ *(120%)*
Thomassen *et al.*, (2008)	Representative conventional and organic Dutch milk production	Cradle-to-farm-gate	1 kg ECM*	Economic allocation Conventional: 91% milk; 8.2% animals; 0.8% exported crops Organic: 90% milk; 6.6% animals; 3.4% exported crops and manure	1.410 *(165%)* 1.480 *(173%)*
Basset-Mens *et al.*, (2008)	Average New Zealand milk production and intensification scenarios	Cradle-to-farm-gate	1 kg NZ milk and 1 kg ECM for comparison	Biological causality (fodder requirement): 85% milk/15% meat	**0.856** *(100%)*

*: ECM = Energy Corrected milk also called fat and protein corrected milk (FPCM)
#: Milk production efficiency is seen as a subordinate goal compared to environment performance because there is a surplus of milk in the region
§: GWP result expressed per kg fat-corrected milk estimated from a result of 10.6 per $10m^3$ fat-corrected milk

Overall, LCA scientists seem to favour more specific references to their studied system when available rather than a very general reference such as IPCC international. For validation purpose, the explicit presentation of key inventory data, such as for N_2O and CH_4 components for GWP, is of major importance to identify potential errors in LCA studies.

Carbon dioxide component

The methods for estimating the different sources of carbon dioxide emissions are overall quite well described. They consist of a mix of well-known references such as Davis and Haglund (1999) for fertilizers, Cederberg (1998) for feed ingredients or more generally Ecoinvent data, and specific references for each context of production including a specific electricity mix per country based on national statistics. In the NZ context, Basset-Mens *et al*. (2008) adapted Ecoinvent data to the NZ conditions and also used specific LCI data for NZ fertilizers based on industry surveys. Although the references are generally described across all studies, the data themselves are missing in most cases but this is of less importance compared to the methane and nitrous oxide components.

Life cycle impact assessment

The global warming potentials (GWP) used are given in all studies, with most based on IPCC (1997) (Cederberg and Mattsson, 2000; Cederberg and Flysjö, 2004; Haas *et al*., 2001; Casey and Holden 2005; Thomassen *et al*., 2008; Basset-Mens *et al*., 2008) and an exception being Williams *et al*., (2006) who use the most recent GWP factors from IPCC (Ramaswamy *et al*., 2001).

Interpretation

Most studies present an interpretation of their results relating the data used to the goal of their study. Studies based on real farm surveys analyse the variability of their results through statistical tests (Cederberg and Mattsson, 2000; Cederberg and Flysjö, 2004; Thomassen *et al*., 2008) and discuss the representativity of their results compared to the typical practice of dairy farms in their country (Cederberg and Mattsson, 2004; Thomassen *et al*., 2008). Mention is also often made across studies of the uncertainty attached to certain key parameters such as N_20 emission factors and the need for more specific data on this pollutant at the national level (Cederberg and Flysjö, 2004; Casey and Holden, 2005; Williams *et al*., 2006). Thomassen *et al*. (2008) discuss the influence of the allocation rules on the results. Williams *et al*. (2006) and Basset-Mens *et al*. (2008) compare their results with some of the already published references selected here. Williams *et al*. (2006) present some sensitivity analyses and comment on the likely uncertainty of their result. However, no studies include an uncertainty analysis. It is worth noting the discrepancy between studies based on real farm surveys, able to explore the variability of their sample of farms, and studies based on a national average farm using national statistics where the concept used is the uncertainty of their prediction for an average scenario and not its variability at a national level.

Tab. 2: Technical description of conventional dairy farm systems across selected LCA studies

	Cederberg and Mattsson (2000)	Cederberg and Flysjö (2004)	Haas *et al.* (2001)	Casey and Holden (2005)	Williams *et al.* (2006)	Thomassen *et al.* (2008)	Basset-Mens *et al.* (2008)
Country	Sweden	Sweden	Germany	Ireland	England + Wales	Netherlands	New Zealand
Origin of data	1 large conventional dairy farm in West Sweden – season 96/97	8 + 9 conventional dairy farms in South-Western Sweden – season 01/02	6 conventional intensive and 6 conventional extensive farms in Allgäu region	National Farm Survey for the years 1997-2001	Milk structural model including all production modes based on statistics and literature#	10 commercial conventional dairy farms	National statistics, big farm samples (Profit watch)
On-farm area, ha, (ha natural pasture)	n.a.	80(10.5)	32.7 – 34.7	n.a.	n.a.	46.7	115(115)
Housing system and herd management	n.a.	Mix of outdoor grazing, solid manure, slurry & deep litter	n.a.	Outdoor (190 – 240 days grazing per year) and slatted-floor	Mix of outdoor, slatted floor, straw bedding…	Stable (slurry and solid manure mentioned)	Outdoor grazing all year round
Milking cows per herd	n.a.	61	Small herds	n.a.	n.a.	81	315
Stocking rate, milking cows/on-farm ha/year	1.6 max by law	About 1	2.2 – 1.9 German LU (= 500 kg liveweight)	n.a.	n.a.	2.13 Dutch LU◆	2.74
Delivered milk, kg/cow/year (and per on-farm ha)	7813 (kg ECM)	8790 (kg ECM) (7410)	6758 - 6390	Estimated at about 4700 from national data in the paper	6534§	7991	3764 (10313)
Pasture DM intake, kg/cow/year	About 300	n.a.	n.a.	Qualitative information: pasture, hay, silage and a bit of concentrates	2458§	*24%§*	4124
Roughage, kg DM/cow/year	2954	n.a.	n.a.		2409§	*40%§*	402
Concentrates, kg/cow/year	1531	2951	n.a.		1269§	*36%§*	0
Total DM intake, kg DM/cow/year*	4785	n.a.	n.a.		6137§	*About 8000*	4526

*: excluding replacement animals; #: data estimated thanks to weighting factors between farm classes provided by the authors; §: Orders given by the authors; ◆: Dutch LU = = yearly phosphate excretion of one milking cow

Tab. 3: Comparison of inventory data for methane and nitrous oxide components reported in the selected LCA studies for milk production at farm gate.

	CH_4 component (emission in kg CH_4/cow/year)		N_2O component	
	Enteric fermentation	Manure management	Direct	Indirect
Cederberg & Mattsson (2000)	Swedish EPA: 155 kg	Not mentioned	IPCC (1997) (3.1 kg N-N_2O/ha or 0.36 kg/t milk)	
Cederberg & Flysjö (2004)	Dairy cattle: 126-130 kg Kirchgessner *et al.* (1991) Replacements: 50 kg, Swedish EPA	IPCC (1997) 8.4 -13.6 kg	IPCC (1997) for manure, N fixation and fertilising sources from soils. IPCC (2000) for housing, slurry spreading	IPCC (2000) for N_2O from ammonia volatilisation and nitrate leaching
Haas *et al.* (2001)	References for climate relevant gases: Crutzen *et al.* (1986); Boumann *et al.* (1991); Kirchgessner *et al.* (1991); Gibbs and Woodbury (1993); Heyer (1994); Patyk and Reinhardt (1997); Rück *et al.* (1997) and Mosier and Kroeze (1998)			
Casey and Holden (2005)	IPCC (1996b): 100 kg/cow in milk EPA (1998): other cattle: 50 kg/stock	For dung deposited: EF from Jarvis *et al.* (1995); EF for dung in milking yard according to Misselbrook et al (2001)	Daily excreta: 0.053 m^3/d (Department of agriculture and rural development Northern Ireland, 2003) + EF according to IPCC (1996b) + other references for housing	Not mentioned
Williams *et al.* (2006)	Method used in national inventory 101-162 kg /cow proportional to DM intake with EF specific per feed	Methods used in national inventory	Methods used in national inventory 163-202 g N_2O-N/cow	Methods used in national inventory
Thomassen *et al.* (2008)	Schils *et al.* (2006), fixed values 113 kg/dairy cow/yr in conventional and 128 kg/dairy cow/yr in organic	Van der Hoek and Van Schijndel (2006)	From soils: Mosier *et al.* (1998); IPCC (2006); From manure management: Oenema *et al.* (2000), fixed values animals/soil	IPCC (2006) N_2O from ammonia volatilisation and nitrate leaching
Basset-Mens *et al.* (2008)	Method used in IPCC-NZ inventory: DM intake x EF CH_4 (21.6 g/kg DM) = 97.8 kg* (110.8)#	Method used in IPCC-NZ inventory: faecal DM x EF for excreta on pastures and from pond 1.3 kg	IPCC-NZ inventory method with specific EF_3 of 0.01 instead of 0.02 in IPCC and with specific fraction leached of 0.07 kg/kg excreted or fertilizer applied Kg on-farm N_2O-N = 6.7 kg/on-farm ha (2.45 kg/cow)	

*: excluding replacements; #: with replacements

Discussion and recommendations

Comparability of studies

Across the seven LCA studies for raw milk at the farm gate, the range of results for GWP appears quite consistent: 0.85 – 1.4 kg CO_2-eq per kg ECM despite the large geographical area covered (Tab. 1). This could be explained by the achievement of a certain level of harmonisation for LCA studies applied to agriculture. It could also be interpreted as a poor degree of specificity in the inventory of each system due to the common use of too general references. For more specific discussion on the differences between results, see Basset-Mens *et al.* (2008).

Part of the difference can be explained by differences in assumptions (FU, allocation rules, GWP_i used etc…) which could be avoided but probably also to intrinsic differences in the typical dairy farm studied across countries. However, this is far more difficult to interpret because of a lack of consistent presentation of these systems across the selected studies. Finally, quite ambiguously, part of this difference or consistency might be due to the quality of the references and methods used for the inventory which is far more complicated to harmonise and improve. As mentioned above, the search for a better accounting of specificities of each production context should be defended against a systematic standardisation which would dramatically reduce the prediction ability of LCA studies.

We do consider that the comparison of independent LCA studies is questionable.

Three major aspects arise from this analysis all requiring collaborative work between specialists of a given agricultural product:

- The need for a more consistent reporting scheme across studies
- The need for a harmonisation process
- The need for the production of specific inventory data at least at a national level

Consistent reporting

A lack of transparency and consistency is noticed in the reporting of LCA studies and it is more marked for published papers with limited length than for reports. To avoid the writing of a full report for each LCA study, key information has to be provided in scientific papers to make them more useful for everyone.

First of all, common definitions for the most important parameters such as feed types, livestock units, feed quality and milk quality must be set up. Intensification must also be defined in relation to the choice of one or a few key parameters. Secondly, the key technical parameters for dairy farms (once consistently defined) must be presented as baseline requirement in any document (report or scientific paper) presenting an LCA study. This list must be discussed but should probably include at least: milk production per cow, stocking rate, total DM intake and DM intake per feed type. Furthermore, methods and inventory data for all components of CH_4 and N_2O emissions should be presented. This is essential to ensure identification of potential errors and consistency of the analysis.

Harmonisation process

The goal and scope of the selected studies are as consistent as possible since papers were selected with this purpose. However, the design of a “typical scenario” of dairy farm for a given context or country is either based on a sample of real farms or on an average dairy farm using national statistics. Both methods potentially have drawbacks. Using a sample of real farms could potentially influence the results favourably, since the farms selected for the survey are the ones showing good and exhaustive data for the LCA study, which is often linked with a better management at all levels and possibly a better eco-efficiency. Conversely, using national statistics to define an average scenario is generally not a sufficient source of data to perform an LCA and several other sources of information have to be used which increases the overall inconsistency of the average dairy farm scenario. For instance in Basset-Mens *et al*. (2008), national statistics were used for milk production while a smaller database was used for key average inputs. This was subsequently demonstrated to have degraded artificially the eco-efficiency of the NZ average farm (results not shown). The design of a typical scenario for a given context is a key aspect of LCA studies for any agricultural products and should be treated as such in a process of harmonisation and improvement.

As commonly observed, several key assumptions could be harmonised such as the definition of the functional unit, including a consistent definition of the product quality (ECM), the allocation rules, the system boundaries and the choice of Global Warming Potentials. A trend to use most specific and most refined data when available can be noticed across LCA studies. The purpose of a harmonisation process should not be to pull the overall level down but where possible and useful to give access to everyone to better data and assumptions. For instance for allocation rules, the generalisation of allocation rules based on a biological causality would be desirable and would require the development of a consensual procedure and equation for this calculation.

The question of harmonisation becomes trickier when considering the inventory step. Producing measurements at a large scale for emissions such as nitrous oxide emissions from soils in different countries requires significant resources, time and effort. Certain people will conclude that the only way to harmonise is to use international references such as those proposed by IPCC. This would have the perverse effect to discourage the countries of producing their own data if a norm was developed in that sense and would cancel the efforts of the countries having their own data. To raise the level, the use of national inventory data should be presented as the favoured option in norms, and only as a default alternative, the use of international default emission factors from IPCC for GWP. LCA studies done in countries with no national data could seem disadvantaged, but this would put pressure on governments from industry and other stakeholders to give scientists resources to produce them.

Regarding the CO_2 components when the major manufactured inputs, such as fertilizers or fuel, are used in a similar context (EU here), a harmonisation process could also be applied. One option could be to use the Ecoinvent data as a reference. In other contexts (such as non-EU), the Ecoinvent data and methods represent a base for developing more specific inventory data, while specific LCA studies are still under development for these products.

LCA studies in general

The analysis and suggestions made for GWP are applicable to all other impact categories except for the inventory step, where general references such as IPCC are not available and where larger differences are incidentally observed across studies (Basset-Mens *et al*., 2008). One idea could be to create an international working group for each product category to define harmonised rules of practice and reporting for LCA studies with a coordinator. Meetings of this group could take place before or after international conferences to reduce the need for specific travel and funding. Most work would certainly be at the inventory stage.

Conclusions

Over fifteen years of application of LCA to agricultural products, the methods used have reached a certain level of harmonisation. However, as described in this review for the GWP of raw milk across countries, significant progress is still required in terms of harmonisation and reporting before being able to reliably compare independent studies. This would require the creation of international working groups per product category and the progressive definition of consistent and harmonised methods and data across all studies. Although the limits of scientific knowledge will always set up the limits of LCA studies, the current and strong need for LCA data standing international scrutiny could steer up scientific research for key knowledge gaps at a global level.

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Sustainable livestock industry: Limitations of LCA methodology

C. Alvarado-Ascencio[1], A. De Schryver[1], H. Blonk[2], M. Vieira
[1]PRé Consultants BV., 3821AD-Amesrfoort, The Netherlands, alvarado@pre.nl.

[2]Blonk Milieuadvies, Gouda, The Netherlands.

Keywords: Meat products, animal husbandry, organic meat, allocation.

Abstract

The Dutch Consumers Organisation commissioned a life cycle assessment study on 10 regular and organic meat products; including lamb, turkey, chicken, cow and pork. The study was performed by PRé Consultants BV and Blonk Milieuadvies.

Economic allocation is used to model the multiple outputs of crop production for animal feed and the slaughterhouse, in which system expansion is not possible. The assessment of meat products is done using two functional units: 1 kg product and 1 euro.

The Impact assessment methodologies Ecoindicator 99 (Goedkoop & Spriensma, 1999) and CML-IA (Guinée *et al.*, 2002) are used to calculate normalised figures for: energy use, land use, climate change, acidification and pesticide use. The study includes also a qualitative analysis of the impacts of replacing nutrients and metals from crop producing countries to cattle keeping countries.

When LCA is used to analyse meat production systems, consistently the lowest environmental impact is associated to the most intensive production systems. The reasons for this are, among others: methodologies often do not account for all pesticides and hormones used, data on specific use of pesticides and hormones is often too complex to collect and animal welfare is not considered.

This paper discusses the consequences of applying economic allocation on the LCA results. Furthermore the paper discusses the limitations of the LCA methodology current status as tool to determine criteria for sustainable livestock industry.

Introduction

Increasingly more attention is given to the environmental impact of the agri-food sector. The study “Environmental Impact of Products” (Tucker et al, 2006), commissioned by the European Union, concluded that a fifth of the environmental impact of the European economy can be associated to the sector “Meat and Meat Products”. This paper explores the possibility of using LCA results as input for policy making towards sustainability.

With the subsidy of the Dutch Ministry of Agriculture, the Dutch Consumers Organisation commissioned a life cycle assessment study on 10 regular and organic meat products; including lamb, turkey, chicken, cow and pork. The study was performed by PRé Consultants BV and Blonk Milieuadvies.

Methods

Economic allocation is used to model the multiple outputs of crop production for animal feed and the slaughterhouse, in which system expansion is not possible. This type of allocation was used because it is suggested as a better option than mass allocation in (Guinée *et al.*, 2002). This type of allocation is consistent with earlier LCA work of the Dutch Consumers Organisation. The assessment of meat products is done using two functional units: 1 kg product and 1 euro.

The Impact assessment methodologies Ecoindicator 99 (Goedkoop & Spriensma, 1999) and CML-IA (Guinée *et al.*, 2002) are used to calculate normalised figures for: energy use, land use, climate change, acidification and pesticide use. The study includes also a qualitative analysis of the impacts of replacing nutrients and metals from crop producing countries to cattle keeping countries.

Data

The main sources of data for this study are the following:

Animal Fed: The databases of Blonk Milieu Advies provided data for the production of animal fed. This database is based in multiple studies on bio fuels and crop production.

Farming: An emission model was developed for this study. This model is based on existing models for the emissions of greenhouse gases (Schils, 2006) and greenhouse gases and Nitrogen emissions (Blonk and Hellinga 2006)

The data sources used to model the production of animal fed and farming are included in Tab. 1.

Tab. 1: Data sources used to model the production of animal fed and farming

Title	Reference
Duurzaam Bier Visiedocument Gulpener	(Aarts, R. en T.J. Blonk, 2005)
Samen met kwaliteit naar de top, Samenvatting van het rapport 'teelt, tafel en traject: de aardappelverwerkende keten',	(Anonymous 2001)
International Fertilizer Industry Association (IFA) 1996-2004,	(Anonymous 2004)
Quick scan milieuvergelijking bietsuiker en rietsuiker,	(Blonk, T.J. 2001 [1])
Monitoring van de duurzaamheidsprestaties van de Nederlandse Varkenshouder.	(Blonk T.J. en C.H. Hellinga, 2005)
Werkdocument broeikaseffect varkenshouderij - analyse t.b.v rekenregels voor de duurzaamheidsmonitor	(Blonk T.J en C.H. Hellinga. 2006)
Milieuanalyse ten behoeve van Milieukeuronderzoek biodiesel.	(Blonk, T.J. 2006)
Zware metalen in de melkveehouderij resultaten en aanbevelingen vanuit het project ‘koeien & kansen’	(Boer, M., Hin K., 2003)
Intersectorale samenwerking in de biologische landbouw, mengvoederproductie met binne- of buitenlandse oorsprong: effect op energieverbuik van mengvoederproductie.	(Bos, J.F.F.P. 2006)
Uitgebreide Energie Studie voor NVM,	(Eijk, J. van, 2005)
data regarding fertilizers use.	(FAOSTAT 200)
Energy Improvement and Cost Saving Opportunities for the Corn Wet Milling Industry,	(Galitsky, C. Worell, E. Ruth, M. 2003)
Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventoris,	
KWIN Akkerbouw en vollegrondsgroenteteelt	
Rapeseed crushing,	LCAfood 2001, Denmark 2001,
Greenhouse Gas Emissions in the Netherlands, 1990-2004	(L.J. Brandes, G.E.M. Alkemade,P.G. Ruyssenaars, H.H.J. Vreuls, P.W.H.G. Coenen),
Biologische landbouw en koolstofvastlegging, analyse van de claims van een Amerikaans veldonderzoek	(Slingerland, S en P. van der Wielen, 2005)
Duurzaamheid van de biologische landbouw,	(Spruijt-Verkerke, J., Schoorlemmer, H., Woerden, S. van, Peppelman, G., Visser, M. de, Vermeij, I. 2004)
Milieu Jaarverslag 2003,	(Suikerunie 2003)
Life Cycle Inventory of Biodiesel and Petroleum Diesel for Use in an Urban Bus,	USDA 1998,
Milieumatenstudie van Margarines, Een oefenproject,	(Vis, J.C., Krozer, J., Duyse, P.J.C. van, Koudijs, H.G. 1992)
Milieuevaluatie van inzet van alternatieve (bio-)brandstoffen in de Gelderland 13 energiecentrale	(Vroonhof, J.T.W. Croezen, H.J. en G.C. Bergsma 2005)
Toepassing van LCA voor agrarische producten,	(Wegener Sleeswijk, A. et.al. 1996)
Toepassing van LCA voor agrarische producten. Deel 4a ervaringen met de methodiek in de case akkerbouw,	(Zeijts, H. van, Reus, J.A.W.A., 1998)

Slaughterhouse: Relatively few information has been published on this subject. Most of the data used in this study was obtained through interviews. Tab. 2 shows the data sources used to models the slaughterhouse systems

Tab. 2 Data sources used to model the slaughterhouse systems

Title	**Reference**
Milieukeureisen voor vleesverwerking als aanvulling op de Milieukeureisen voor varkens	(Blonk 2001 [2])
Verwaarding van nevenstromen uit de pluimveeslachterijsector,	(Bolck, 2003.)
Gommen en zetmeel als alternatief voor gelatine	(Hollering, P)
Visie op de varkenskolom	(Hoste, R., Bondt, N., Ingenbeek, P. 2004,)
Data regarding packaging use at the slaughterhouse in Beilen	(Laurus 2007)
Energie in de varkensketen	(Kramer, K.J., et al, 2006.)
Processing and marketing non-meat products from livestock	(Oberthür, 2002.)
Economische berekeningen aan huisverkoop van biologisch rund- en varkensvlees	(Puitser, L.F., Hoste, R.J 2005)
Statistics PVE 2006	
Interview with A Tuit en M van Gogh	
Interview with Dhr Marcelis.	
Prijsontwikkeling in de rundvleesketen,	(Vlieger, de J.J., Bolhuis, J., 2002)

Retail: The data for retail and central slaughter was provided by Super de Boer (Dutch supermarket chain)

Background data: Data for the production of fertilizers, packaging materials, energy production, transport, fuels, etc., was taken from the Ecoinvent database.

Results

When LCA is used to analyse meat production systems, consistently the lowest environmental impact is associated to the most intensive production systems. The reasons for this are, among others: methodologies often do not account for all pesticides and hormones used, data on specific use of pesticides and hormones is often too complex to collect and animal welfare is not considered.

De energy use of regular meat products is determined by the energy to produce the fertilizers needed to cultivate the animal feed. For organic products the energy use is dominated by the consumption of fuels and electricity at the farm. The scores for electricity is strongly dependant of the allocation procedures.

The results for regular and organic beef and lamb give in general the highest scores per kg meat. These systems score especially high climate change scores. This is a result of methane emissions associated to cows and lambs. Also on land use the scores for these systems is higher, however it must be considered that in this case most of the land use is a relatively environmentally friendly use (meadows) which cannot be directly compared with the land use associated to the production of crops.

For beef systems which consider the production of milk the impacts on climate change and land use are comparable to other systems as chickens or pork. This is due to the allocation of the impact between the meat and the milk. Fig. 1, Fig. 2 shows respectively the results for climate change and land use for the functional unit of 1 kg of meat. Fig. 3 and Fig. 4 show a comparison of the results for climate change for the functional units of 1 kg and 1 euro respectively.

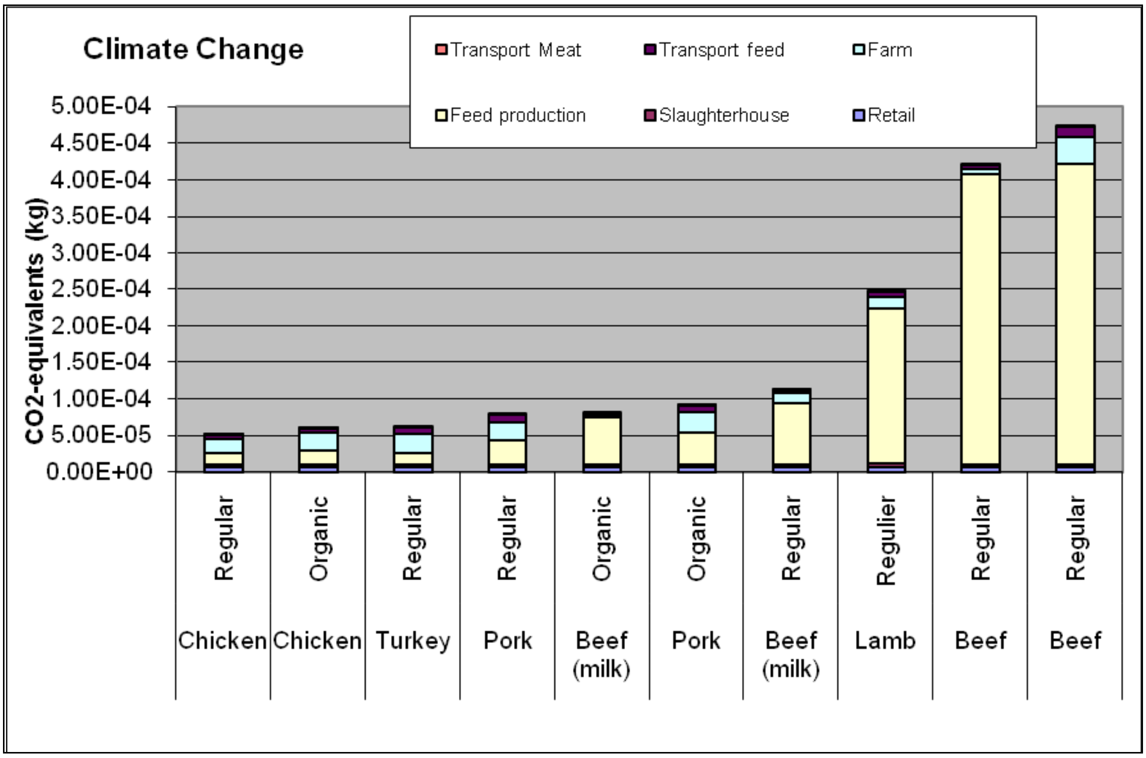

Fig. 1: Climate change expressed in Kg CO_2 equivalents. EcoIndicator 99.

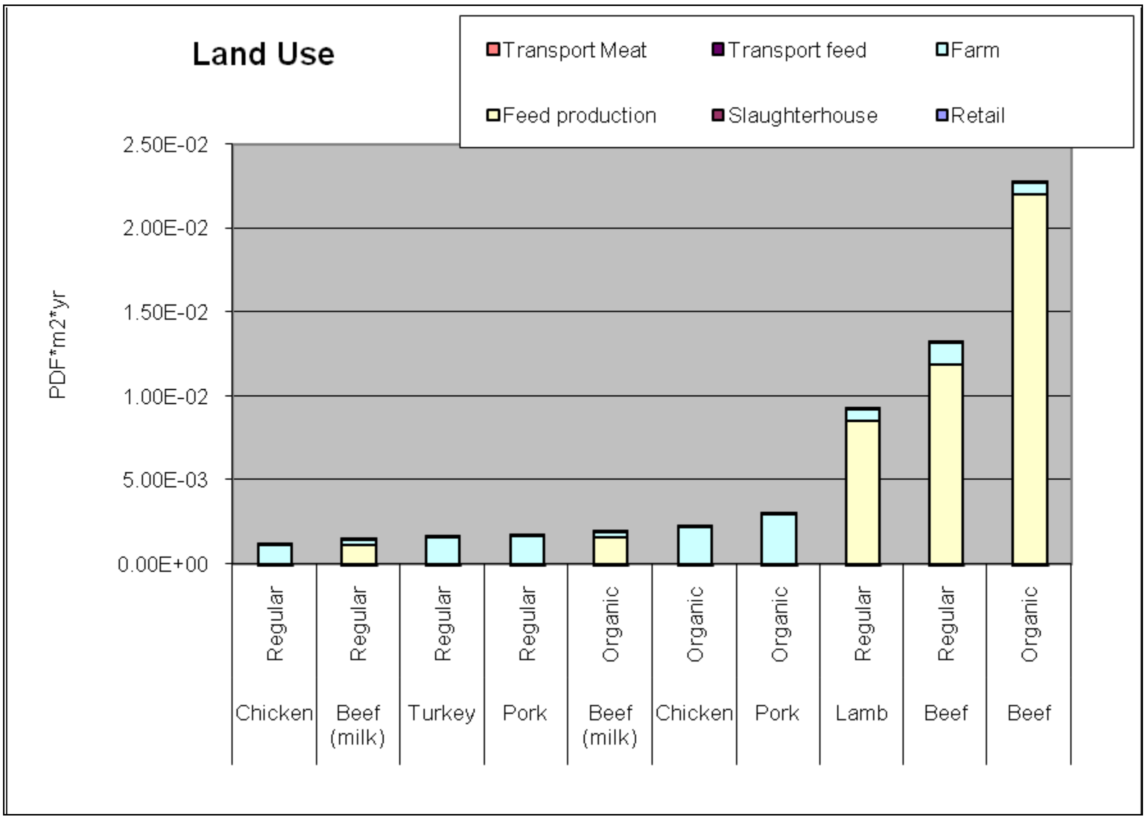

Fig. 2: Land Use expressed in Fraction of potentially disappeared species. EcoIndicator 99.

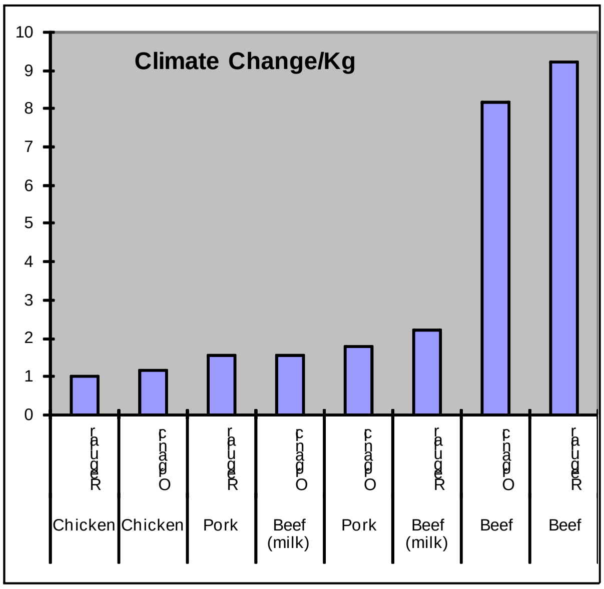

Fig. 3: Left. The amount of CO_2 Equivalents along for the production of 1 Kg of meat. Ecoindicator 99 method. The values on axis y are normalised values relative to the lowest score.

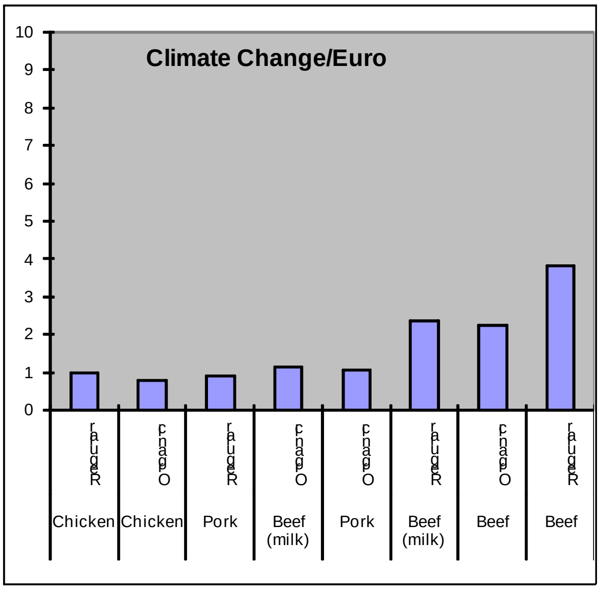

Fig. 4: Right. The amount of CO_2 Equivalents along for the production of 1 Euro of meat. Ecoindicator 99 method. The values on axis y are normalised values relative to the lowest score.

Discussion

The results presented in this study are subject of consideration. In many cases more data and further analysis is required. As in the case of allocation, emissions of heavy metals and land use.

In first place it is necessary to make difference among local and global impacts. In the first category we find acidification/eutrophication, and emissions of heavy metals. In the second category we find energy use and Climate change. In this sense land use and the use of pesticides is an intermediate category since these impacts are actually local however the available impact assessment methodologies do not allow for a regional analysis. In the case of land use a more detailed characterisation in the impacts assessment method may be required to assess land use along the life cycle. In this study land use is assessed in a worldwide way.

In second place the effect of the functional unit is important. If one euro is used as functional unit the differences among meat sources are smaller than if the functional unit is 1 kg. This raises the question whether the value of products has relation to their environmental impact. To certain extent this is the case, since the rank of products remains for both functional units. Animal fed conversion is a critical factor for both production costs and environmental impact. The costs associated to land use are also associated to the production costs and environmental impact. In other words intensive production with little land use is beneficial for the economy and the environment. This has little consideration with other effects which are not visible as the use of hormones, pesticides, and animal welfare.

The use of economic allocation has a strong influence in the results and it raises the question whether it is a fair approach to the livestock industry. Many of these by-products have a very low value when at the slaughterhouse; however they give place to high value products for the pharmaceutical and food industries. This is the case of products like gelatine produced out bones or globulin, which is produced out blood. Fig. 4 shows that meat is only one of the many products of the livestock industry. The Depending on the animal 30 to 60% of the total mass may end up as by-products. This situation favours cows, porks and lambs over chicken and turkey. Unfortunately the timeframe of the project did not allow for a more complete analysis of the value of the byproducts of the livestock industry. This is the main limitation of the present study and is an indication of the possible overestimation of the environmental impact of beef, pork and lamb meat over chicken and turkey.

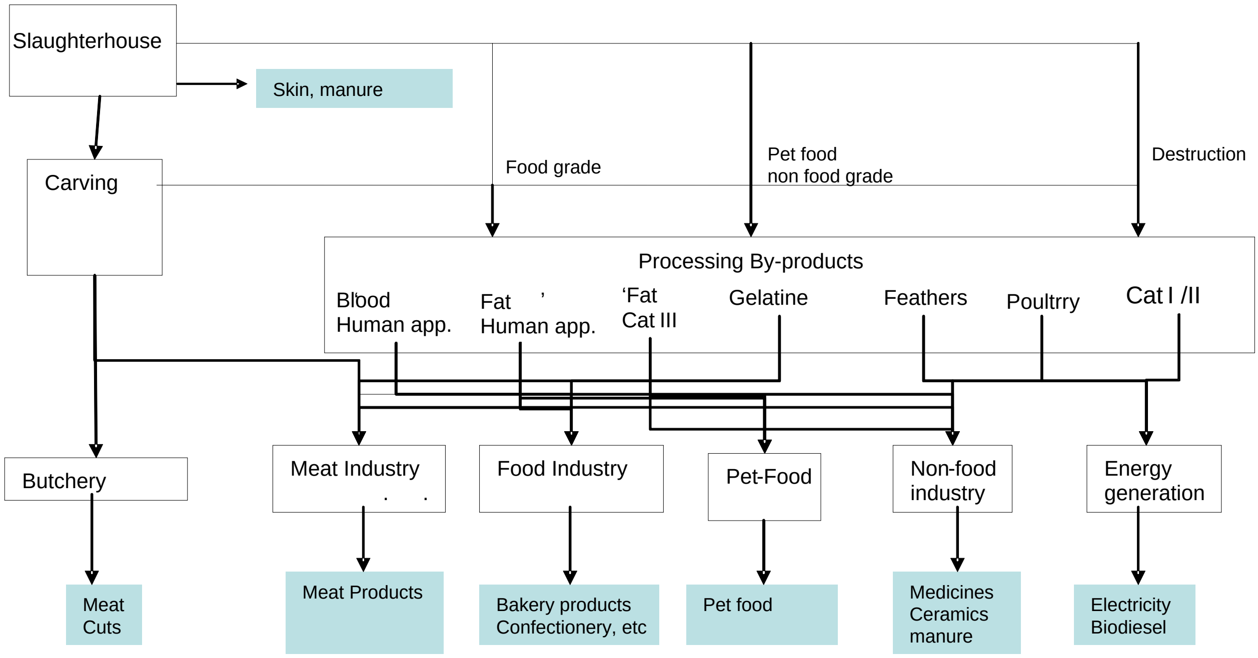

Fig. 5: Products and by-products of the slaughterhouse

Conclusions

Due to the shortcomings on the allocation of impacts to the different products of livestock industry, it is not possible to extract concrete results on the comparison of different meat production systems. In all impact categories further analysis is necessary. The systems for regular and organic beef and lamb have the highest scores per kg meat for climate change. This is also the case per euro product. For the same impact category pork, chicken and turkey yield the lowest scores. Beef systems that consider the production of milk yield scores which are comparable to those of chicken. This is strongly dependant of the allocation procedure.

Organic beef systems have the lowest energy requirements while regular beef systems have the highest energy requirements.

On land use beef, organic pork and chicken meat systems have the highest scores. This is factor two higher than in regular systems. While in terms of LCA this is associated to a higher environmental impact, a higher score on land use also means a positive effect on welfare.

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Investigating variation and uncertainty in agricultural production systems: examples from Australia

D.R. Farine[1*], D. O'Connell[1], T. Grant[2], P.J. Thorburn[3]
[1] CSIRO Sustainable Ecosystems, GPO Box 201, Canberra, ACT 2001, Australia
[2] CSIRO Marine & Atmospheric Research, PMB1 Aspendale, Vic. 3195, Australia.
[3] CSIRO Sustainable Ecosystems, Queensland Bioscience Precinct, 306 Carmody Road, St Lucia, Qld 4067, Australia.
*Corresponding Author: Damien.Farine@csiro.au

Keywords: LCA, Biofuel, Sugar, Wheat

Abstract

The importance of reducing greenhouse gas emissions is becoming globally recognised, and the replacement of oil with biofuels is one proposed method of achieving this goal. The emission profiles of biofuels are of critical interest as a key aspect of their sustainability and LCA has been successful in helping characterise these. However new technologies, increased understanding of the role of nitrous oxide emissions, novel feedstocks and an emerging capacity to understand variation in farming systems are pushing a need to revisit existing knowledge. Using LCA, variation was explored within and between regional wheat and sugar agricultural systems, resulting in differences in emissions of up to 87% in sugar production and 64% in wheat production. Combining LCA with an agricultural systems simulator allowed an investigation into model uncertainty due to the choice of emission factor in sugar systems, which were found to alter the results by up to 72%. Such analyses may prove useful in the rapid assessment of future biofuel feedstock emissions and help to accurately report on key sustainability parameters.

Introduction

A global recognition of the importance to reduce greenhouse gas (GHG) emissions, coupled with a need to improve energy security, is driving a move to reduce reliance on oil. Biofuels are one of a range of proposed alternatives, however the biofuels industry in Australia is currently small, supplying less than 0.5% of the total transport fuels (O'Connell *et al.*, 2007). There is continued interest in expansion of the industry, though there are challenges to doing this in a sustainable manner. Whilst currently based around waste starch and C-Molasses for ethanol, and used vegetable oil and tallow for biodiesel, significant industry growth would require new feedstocks sources because the supply of current feedstocks is reaching its limit. Demonstration of sustainability credentials supported by robust science is an important step for industry expansion.

There are many dimensions to sustainability assessment. Lifecycle Assessment (LCA) is one useful approach to quantifying particular aspects of sustainability, and comparing the merits of contested options. GHG emissions of biofuels have long been of critical interest to policy makers as a key aspect of sustainability, and have been fundamental to the argument for government support. The GHG emission profiles for a range of standard first generation biofuels in Australia were characterised using LCA (Beer *et al*. 2001; CSIRO, ABARE and BTRE 2003), and have been used as policy benchmarks. The analyses modelled GHG emissions of biofuel blends, taking account of variation in a limited set of blends (B5, B20, B100 and E10), and comparisons made to conventional fossil fuels. These only partially accounted for the different feedstock categories and conversion technologies under consideration in the early 2000s, and did not take into account any detailed knowledge of the production systems. Thus, average values were used for management practices, including fertiliser and other inputs, environmental variation, and in general soil carbon was held in equilibrium.

These greenhouse gas profiles for biofuels need revision due to recent developments in:

- the emerging capacity to link agricultural simulation models to LCA which now allows exploration of range of variation in the production systems, including management and environment interactions, and the capacity to partition 'green' and 'blue' water use (Falkenmark *et al*., 1998);
- scientific understanding of the extent and role of nitrous oxide emissions (e.g. Crutzen, 2008) and our ability to model them for some systems;
- second generation technologies which enable use of a range of new lignocellulosic feedstocks (e.g. Hamelinck & Faaij, 2006; Warden & Haritos, 2008); and
- novel feedstock sources for oils (such as *Pongamia pinnata* as an oilseed tree, or algae) (Scott *et al.*, 2008).

This paper reports on the progress in linking agricultural simulation models to LCA as an exploratory tool to better understand uncertainty and variation in the feedstock production end of the biofuels value chain.

Australian wheat production systems

Australian wheat is produced across three agronomic regions, the southern and western regions (which have similar practices of continuous cropping across years and are here treated as one), and the northern region (where a fallow year is undertaken between each rotation of wheat for water management). Altogether, approximately 13 million hectares of wheat are grown in these regions combined, representing 3% of total Australian land area, and producing 25 million tonnes of grain annually (ABS, 2008).

Farine *et al.* (2008) performed Lifecycle Assessment on ten typical Australian wheat farms. Firstly northern and southern Australian systems were compared where wheat is produced farm-wide each year (southern) vs. wheat is produced on 50% of the land in rotation with a year-long fallow (northern).

Secondly, variation within the southern Australian wheat growing region was investigated with the following rotation options:

- wheat is produced farm-wide each year using conventional tillage vs. wheat produced farm-wide each year using minimum tillage (no or zero till),
- wheat is produced farm-wide each year vs. wheat is produced on 50% of the land in rotation with a crop or pasture legume (both using minimum tillage),
- wheat is produced farm-wide each year vs. wheat is produced on 50% of the land in rotation with canola (rapeseed in Europe) where 10% of the canola is used for biodiesel (both using minimum tillage), and
- wheat is produced farm-wide each year vs. wheat is produced on 40% of the land, with 10% as a dedicated biodiesel crop (canola) and 50% in a legume rotation (all using minimum tillage).

Each system has distinct management options, including fertiliser, pesticide, herbicide and machinery inputs which affect the emission profile of the farming system. The National Greenhouse Gas Inventory Committee's (2006) standard emission factor of 0.3% of N applied for dryland cropping was used to calculate nitrogen emission.

Variation in wheat production emissions between northern and southern Australian regions

Comparisons of the two main wheat growing areas in Australia showed the variation caused by their environmental differences (Fig. 1). The northern Australian region, due to summer rainfall patterns and summer cropping, must include a year-long fallow rotation for each wheat rotation. Thus the impact of each hectare farmed (where one modelled hectare is comprised of 50% wheat and 50% fallow) was 29% lower than the southern systems due to the low-input nature of fallow rotations. However, when a functional unit of per tonne of production was used, the southern system had 22% less emissions since all of the inputs went directly into growing the wheat, whilst northern systems had some inputs into the fallow rotation (tillage or sprays) in addition to those put on the crop.

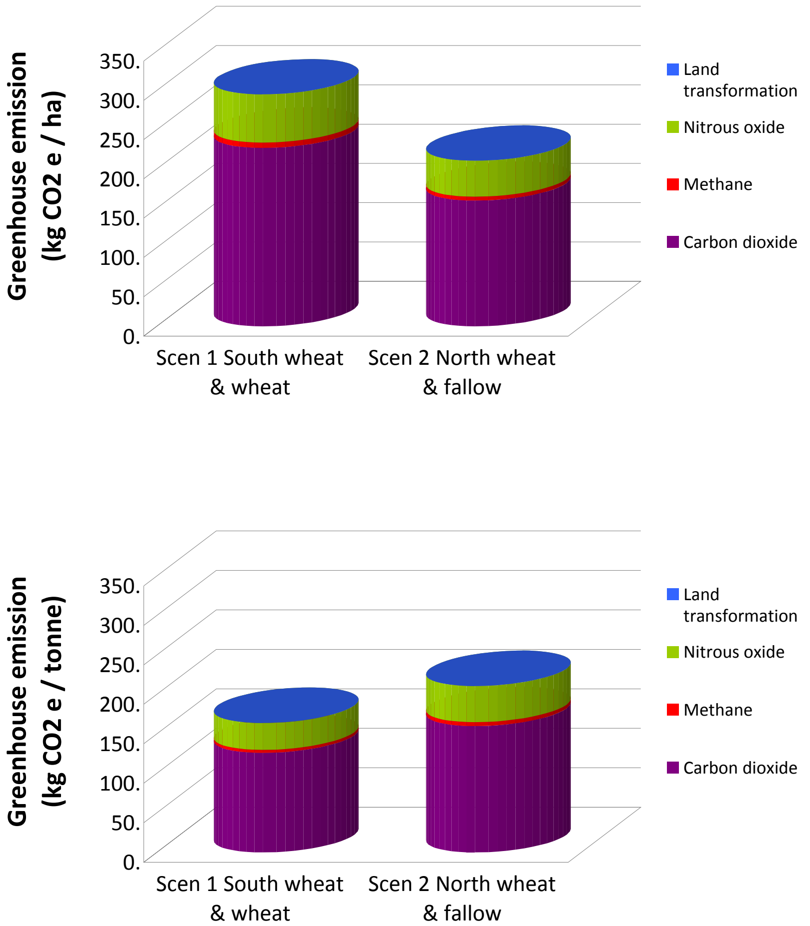

Fig. 1: Regional variation in greenhouse gas emissions between southern and northern wheat farming systems in kg CO_2-equivalent per hectare (top) and per tonne (bottom) (derived from Farine *et al.*, 2008).

Performing this comparison highlights the different perspectives of viewing farms as enterprises based on the use of a given area of land, and production systems producing grain for local use or export. Both of these are important for planning and optimising the greenhouse emission from Australian agriculture. It may be possible to increase production with similar or lower greenhouse emissions per tonne, but this may lead to increase total greenhouse emission within that region or in Australia as a whole.

Variation in wheat production emissions within the southern Australian region

In southern Australian wheat production systems, a number of rotational and management decisions are possible, leading to variations in the GHG emission profiles (Fig. 2). The difference between conventional and minimum tillage alone was approximately 8%, whilst introducing nitrogen-fixing legume rotations reduced GHG emissions by 39 to 56% per hectare, or 26-29% per tonne of grain production due to reduced fertiliser application, though this also reduced wheat production by 39%. Using a rotation of wheat and canola, where a percentage of the canola was used for biodiesel (10% in this case), had some GHG emission savings due to the biodiesel offsetting fossil diesel and managing a large crop of both wheat and canola. Finally, a system aiming to drastically reduce emissions managed savings of up to 64% per hectare, or 27% per tonne of production by combining methods for reducing fertiliser application (using 50% of the crop area under legume), growing canola for biodiesel production (10% of the crop area as a dedicated biodiesel crop), and the remaining 40% under wheat production.

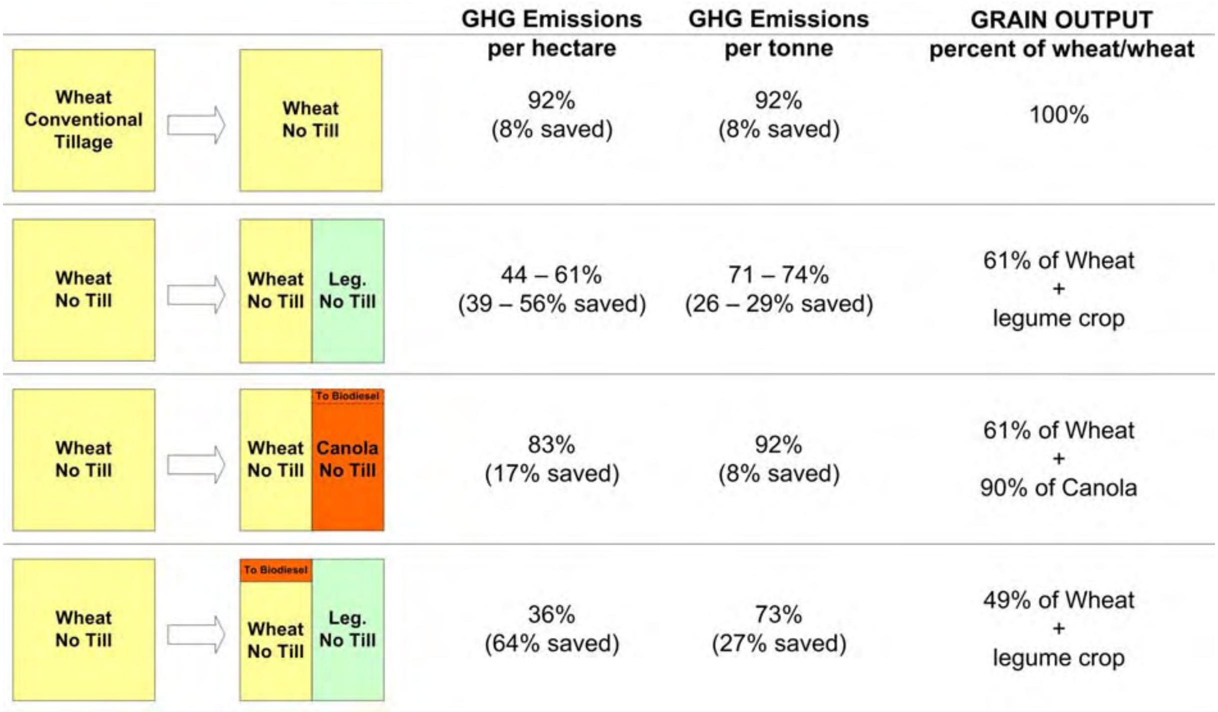

Fig. 2: Synthesis of the results from Farine *et al.* (2008) showing the greenhouse gas emission savings of various management options and the production costs incurred by each.

The results in Fig. 2 are important if a greenhouse gas signature is required for a biofuel such as ‘wheat to ethanol’. This GHG signature would therefore depend on the farming system in which the wheat was produced and, as shown in Fig. 1, also on the metric against which the emissions are reported. The variation reported in Fig. 2 may occur at a paddock-scale, further increasing the difficulty of capturing accurate emission profiles for a given fuel.

Australian sugar production systems

Sugar is grown primarily in north-eastern Australia on approximately 400,000 hectares producing 38 million tonnes of sugar cane, or approximately 5.2 million tonnes of raw sugar (ABS 2008; Sugar Australia 2008). An exploratory study by O’Connell *et al.* (2008) compared the emission profiles of three different sugar growing regions:

- Tully, northern Queensland
- Burdekin, central Queensland
- Maryborough, south-eastern Queensland.

In Tully and Maryborough, sugarcane is commonly harvested without burning and the trash is spread on the ground following harvest. In the Burdekin, trash is generally removed through pre and post-harvest burning.

Variation within the Tully region was also explored by modelling the effects of time since clearing of paddocks. Three different histories were used:

- long-term sugarcane production, soil carbon close to equilibrium,

- counting soil carbon run-down from immediately after clearing took place (over 44 years), and
- counting above-ground biomass loss from clearing and soil carbon rundown (over 44 years).

Finally, model uncertainty (Huijbregts 1998) was investigated by re-analysing the three main regional systems (above) using nitrous oxide (N_2O) emissions modelled explicitly in an agricultural production system simulator (APSIM, Thorburn *et al.* 2008) rather than estimating emissions using the National Greenhouse Gas Inventory Committee's (2006) standard emission factor of 1.25% of N applied to calculate nitrogen emission.

Variation in sugar production emissions between growing regions

The main source of variation in sugar production systems between regions was due to the interactions of management and environment. For example, wetter, warmer areas such as Tully had higher yields but also a higher use of pesticides and tractor fuel for controlling weeds. Nitrous oxide emissions were related to the different applications of nitrogen fertiliser in the different regions. These systems were further affected by irrigation (Burdekin and Maryborough), as well as decisions made on burning the trash, which caused the high methane emissions in the Burdekin paddock.

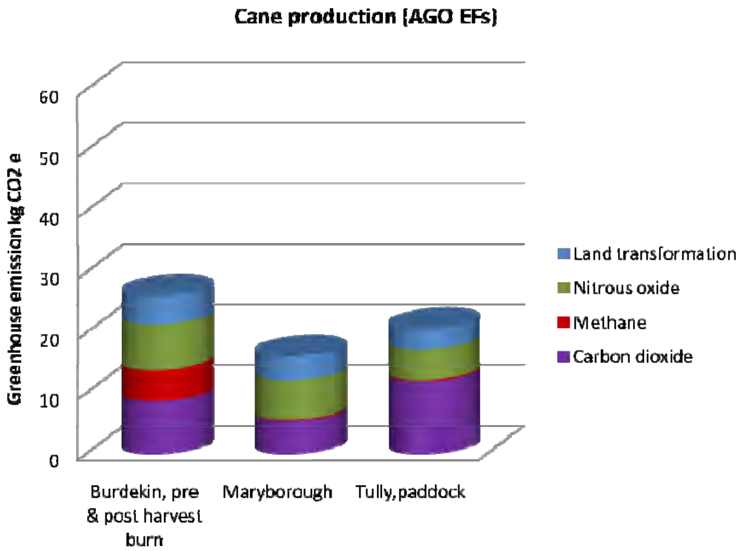

Fig. 3: Inter-regional variation in the production of sugar (O'Connell *et al.*, 2008)

The results of this analysis (shown in Fig. 3) showed variation in emissions of sugar production was caused by regional differences in the production systems used to grow sugarcane. There is scope for reducing emissions by varying management practice. For example, ceasing to burn trash would markedly reduce GHG emissions from sugarcane production in the Burdekin region, whereas reducing nitrogen fertiliser rates in the Maryborough region may also reduces GHG emissions.

Variation in sugar production emissions within region due to historical land-use

The effects of clearing rainforest on the greenhouse gas emission profile of sugar production were explored using three systems with different historical land-uses within one sugar production area (Tully, Queensland). The variation from historical land-use alone (important in carbon accounting) can be up to 87% as shown in Fig. 4. In this example, a cleared paddock (60+ years) with soil carbon near equilibrium was compared to a paddock farmed immediately after clearing with soil carbon rundown amortised over 44 years, and a paddock cleared for sugar production where soil carbon and above-ground biomass carbon losses were amortised over 44 years (O'Connell *et al.* 2008). In the third case, it would take 59 years of producing ethanol from C Molasses in order to attain carbon neutrality from petrol offsets. Therefore any argument supporting net GHG benefits for ethanol production from sugar in newly cleared land would be highly questionable.

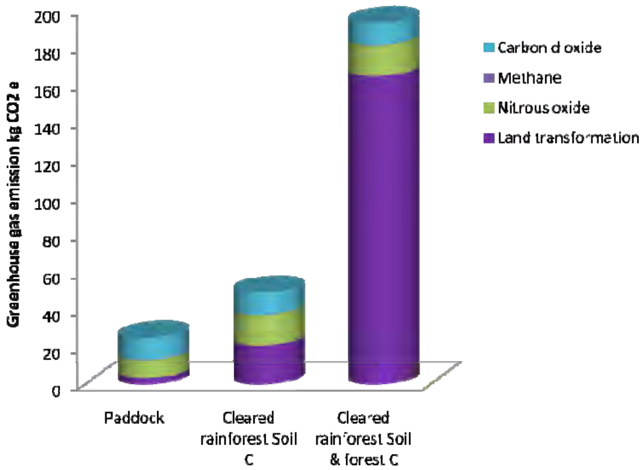

Fig. 4: Variation in greenhouse gas emissions profiles due to time since clearing (from O'Connell *et al.*, 2008)

Uncertainty in sugar production modelling due to N_2O emission factors

In Kyoto carbon accounting, standard factors are used for nitrous oxide gas emissions from production systems. Crutzen (2008) showed that these are vastly underestimated in many systems (where uncertainty ranges between one-third to three times the value of the emission factor). Thorburn *et al.* (2008) are developing a nitrous oxide emission capability in the soil nitrogen module of the well-established APSIM software. Early implementation of this capability in sugar production has shown differences in overall emissions profile of the three sugar producing regions when the emissions factors were modelled this way (O'Connell *et al.*, 2008). A comparison of Fig. 3 with Fig. 5 showed a differences of up to 72% (in Maryborough) between the two systems when using N_2O emissions modelled in APSIM (ranging between 3 and 7%) compared to the standard industry emission factor of 1.25% used by the National Greenhouse Gas Inventory Committee (2006).

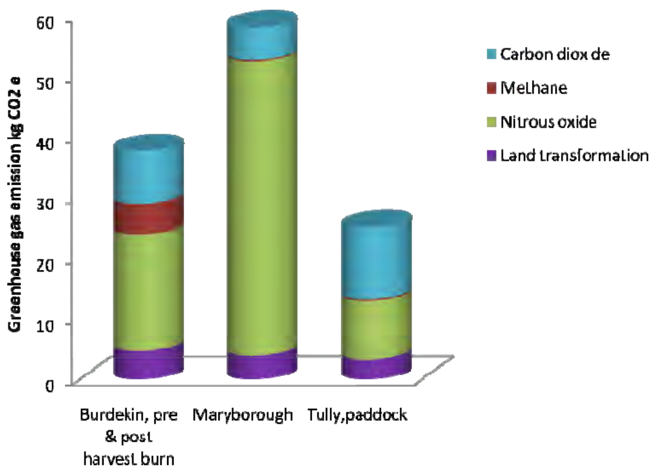

Fig. 5: Greenhouse gas emissions profile of the three modelled sugar growing regions using nitrous oxide emissions factor simulated in APSIM (O'Connell *et al.*, 2008)

Discussion

The presented studies (Farine *et al.*, 2008; O'Connell *et al.*, 2008) reported the early results of using simulation modelling combined with LCA to understand variation and uncertainty in agricultural production systems. The GHG emission profiles of wheat systems showed that the intra-region variation based on management decisions (rotation choices) was greater than the inter-regional variation (northern v southern farming systems). The GHG emission profiles for sugar systems were highly variable within and between growing regions, depending on which land transformations were accounted for, and especially due to the interactions between management and the environment.

Uncertainty of N_2O emission factors are a challenge for the lifecycle analyst. The accuracy of representation of emission factors differs in various agricultural systems. The study by O'Connell *et al.* (2008) reported that N_2O emissions may be higher (at 3 to 7% of N applied) than reported using the standard factor (1.25%; National Greenhouse Gas Inventory Committee, 2006). In contrast, a recent study by Barton *et al*. (2008) reports that N_2O emissions from field trials in wheat systems of south-western Australia are 0.02%, below the National Greenhouse Gas Inventory Committee's (2006) factor of 0.3% for dryland cropping, and much below IPCC's (1997) default value of 1.25%. Thus, given the importance of N_2O with 310 x Global Warming Potential of CO_2 (IPCC, 2007), more work is required to quantify N_2O emissions across the spectrum of farming systems. Until such time that the range and probability of N_2O emission factors are known for each system and region (allowing monte carlo simulation to explicitly state the uncertainty; Huijbregts 1998; May & Brennan 2003) the impact assessment scores should be reported using all representative emission factors (May & Brennan 2003).

The biofuels industry is growing rapidly across the globe and increased demand may lead to new production systems. As well as the introduction of lignocellulosic conversion technologies leading to new feedstocks, many promise the answer lies in speciality feedstocks, such as oilseed trees (*Pongamia pinnata* in Australia; Scott *et al*., 2008). Previous analyses of these specialised species are problematic as they are often based on overseas performance in environments quite different to Australia. They do not consider the large climatic variation in Australia, and environmental sustainability implications of each growing region. Further, the assumptions used for estimating these new feedstocks are often based on the use of 'marginal land' as an analogue for transfer of production potential - when the drivers of 'marginality' of land may be very different on the Australian continent compared to the locations from where their production potential results are drawn. Simulation models and Lifecycle Analysis may be useful to guide investment of field and research effort for these new and emerging feedstocks.

LCA is helping guide knowledge and future policy with regards to environmental issues and drivers in Australia. It was greatly enhanced, however, when linked with production systems simulators. This allowed for rapid investigations to be made of a number of scenarios, enabling better understanding of the main functional parameters and their variation. While useful for initial sustainability assessment and system design for biofuels, this work may be useful for other value chain work, such as regional product differentiation for an increasingly discerning consumer market. Finally, balancing the complexity of variation with the simplicity required for policy implementation will be challenging when moving beyond 'single value' emission reductions. This raises the question of how these different production systems might be handled in a policy context, or if any 'track and trace' style of sustainability certification of the feedstock were required. There is a great deal of research still needed to address these issues.

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Environmental evaluation of cow and goat milk chains in France

C. Kanyarushoki [1], F. Fuchs [1], H.M.G. van der Werf [2]
1 BTPL, La Futaie, 72700 Rouillon, France, claver.kanyarushoki@rennes.inra.fr, f.fuchs@btpl.fr
[2] INRA, UMR 1069 Sol Agro et hydrosystème Spatialisation, 35042 Rennes, France

Keywords: cow milk, goat milk, butter, energy use, dairy production chain, LCA

Abstract

Stakeholders of the Poitou-Charentes (central western France) milk chain work together to analyse the environmental impacts of regional dairy chains to identify improvement options. Thirteen cow farms and six goat farms of the pays Thouarsais were analysed. Per 1000 kg milk, Thouarsais cow milk had higher impacts than Bretagne cow milk; per ha of land occupied its impacts were similar or less. Per 1000 kg milk, Thouarsais goat milk had higher impacts than Thouarsais cow milk, per ha of land occupied its impacts were largely similar. A preliminary analysis on the effect of the exclusion of cash crops from the Thouarsais dairy farms suggested that the use of economic data to allocate impacts to milk, animals and cash crops was not the best option for these farms, some of which make more money from crops than from milk. For all impacts except energy use, farm operations was the stage contributing most to overall impacts of the cow milk chain, impacts of farm inputs came second. For the post-farm dairy chain, impacts associated with the transport of products to retailers were more important that those of the dairy plant. These results will be analysed with project partners to identify the most promising improvement options for all stages of the milk chain.

A country producing almost 360 different types of cheese cannot die
Winston Churchill in June 1940

How can you govern a country which has 246 varieties of cheese?
Charles de Gaulle (from Les Mots du Général, Ernest Mignon (1962))

Introduction

Although Winston Churchill and Charles de Gaulle seem to differ on the number of varieties of cheese existing in France, they agree on a more fundamental point: cheese is at the heart of French identity. The rich biodiversity of cheese and other dairy products found in France contributes to the pleasures of a varied and healthy diet, but also to the regional and national economy.

In the Poitou-Charentes region in central-western France, the production and transformation of cow and goat milk is of major economic importance. Dairy farms contribute to shaping this region's attractive bocage landscape, a harmonious mix of hedges, pastures and annual crops. Its dairy products are sold well beyond the regional and national borders and their quality enhances the image of this region. A varied group of stakeholders of the Poitou-Charentes milk chain have recently decided to work together in a research and development project called PaRMEELI (http://www.btpl.fr/page.php?r=4&p=44), that aims to analyse the environmental impacts of regional dairy chains in order to identify improvement options across the entire milk chain from the farm and its inputs up to the transport of dairy products to retailers. Stakeholders in this project are organisations involved in dairy farm development, promotion of energy conservation, training in milk technology, agronomic research, technical advice to farmers, and regional sustainable development. This paper presents the first results obtained in the PaRMEELI project, which was launched in February 2007.

Thomassen *et al.* (2008) compared several LCA studies of cow milk production up to the farm gate. They found impacts per ton milk of 2.8 – 10.5 kg-eq. PO_4 (eutrophication), 10 – 22 kg-eq. SO_2 (acidification), 900 – 1500 kg-eq CO_2 (climate change), 1.2 – 5.0 MJ (energy use) and 1300 – 2900 m^2 $year^{-1}$ (land occupation). A number of LCA studies of production and consumption of cow milk and cow milk products have been reviewed and summarised by Foster *et al*. (2006), revealing that the primary production stage (up to the farm gate) contributed most to all impacts considered. Berlin (2002) carried out an in depth LCA of cheese, showing that across the cheese production and consumption system primary production contributed 99% to eutrophication and acidification, 94% to climate change and 69% to energy use.

Methods

We analysed dairy chains using life cycle analysis (LCA). This study was conducted in a pilot area in Poitou-Charentes, the “Pays Thouarsais”, hosting around 140 dairy cow farms and 70 dairy goat farms. Our work focused on specialised dairy farms, i.e. those hosting no other animal species than either cows or goats: 54 cow farms and 41 goat farms. Among these, 13 cow farms (Cow T) and six goat farms (Goat T) were analysed, and compared to a reference group of 46 dairy cow farms in the Bretagne region in western France (Cow B).

LCA calculations at the farm level were performed with a Microsoft® Excel-based tool called EDEN (van der Werf *et al*., in review). For each farm, EDEN estimated emissions of CH_4, CO_2, NH_4, N_2, N_2O, NO, NO_2, NO_x, SO_2, NO_3, PO_4, Cd, Cu, Ni, Pb, Zn, as well as use of non-renewable energy and land occupation. Estimated CO_2 emissions do not include the effects of increase or decrease of carbon stocks in soil. Based on this inventory, EDEN calculated potential impacts for eutrophication (EU, kg PO_4 eq.), acidification (AC, kg SO_2 eq.), climate change (CC, 100-year horizon, kg CO_2 eq.), terrestrial toxicity (TT, kg 1,4-DCB eq.), non-renewable energy use (NR, MJ), and land occupation (LO, m^2 $year^{-1}$). EDEN distinguishes “direct” impacts that originate on the farm site itself from “indirect” (off-farm) impacts associated with the production and transport of inputs to the farm. Impacts were compared among farms by standardizing them to two functional units: a) 1 tonne of fat- and protein-corrected milk (FPCM) sold and b) on-farm plus estimated off-farm hectares utilised. For the functional unit *1 tonne of FPCM sold* total sales data for milk, livestock, and crop products are used to perform an economic allocation, which estimates the proportion of total impacts due to each of these three product types.

EDEN applies a cradle-to-farm-gate analysis, meaning that the system evaluated consists of the farm and its main inputs, but that products are no longer part of the system once they leave the farm. Non-agricultural parts of the farm such as the farmer’s house, woodlands and forests are not included in the system, nor are construction of farm buildings, farm roads and drainage networks. The use of disinfectants, detergents, antibiotics, hormones and other medication is not considered, due to lack of data concerning the production and environmental fate of these inputs. For pesticides, non-renewable energy use for production and supply is considered, but impacts associated with the use of pesticides (toxic effects) are not considered, due to lack of appropriate characterisation factors. In the framework of this project a dedicated version of EDEN for the evaluation of dairy goat farms was developed.

Regarding the post farm dairy chain for cow milk, our results are based on data for a dairy plant in Poitou-Charentes, which transforms milk into butter, crème fraîche and skimmed milk. Hypotheses on the emissions and resource use associated with the on-farm production of the milk that is transformed in this dairy were based on: a) average data for the milk from the 13 Cow T farms, b) average data for the milk of the two Cow T farms with lowest values for NE per 1000 kg FPCM, c) average data for the milk of the two Cow T farms with highest values for NE per 1000 kg FPCM. We divided the cow milk chain originating in the Pays Thouarsais in four stages: 1) production and delivery of farm inputs, 2) farm operation, 3) transport of milk to and operation of the dairy plant and 4) post-plant transport of products (butter, crème fraîche and skimmed milk). Construction and maintenance of the dairy’s buildings and equipment were not included in the system. The use of energy carriers (electricity, fuel oil), packaging materials and chemicals were considered. Data for these processes and for transport were from Ecoinvent v2.0. Temporal coverage was a period of one year, corresponding to the period used in the bookkeeping for the farms and factory. Life cycle impacts assessment methods used were: CML 2001 version 2.04 and Cumulative Energy Demand version 1.05, as implemented in SimaPro 7.

Participative LCA approach

The project involves a wide range of stakeholders associated with Poitou-Charentes cow and goat milk production and transformation chains. LCA plays a central role in realising the objectives of the project; however, many of the partners involved in the project were neither familiar with the LCA approach, nor convinced of its relevance. In order to prevent a "top down" LCA, in which most of the partners would be mere data providers, we decided to experiment a "participative" LCA. This involved frequent meetings in which preliminary LCA results were presented, need for additional data collection at the farm and factory level was discussed, and all partners were involved in deciding on subsequent project stages.

Results

Characteristics of dairy farms

Tab. 1: Mean values for characteristics of dairy farms, Bretagne cow farms (Cow B, n = 46), Thouarsais cow farms (Cow T, n = 13) and Thouarsais goat farms (Goat T, n = 6).

Characteristic	Dimension	Cow B	Cow T	Goat T
Farm structure				
Useable Agricultural Area (UAA)	ha	59	161	120
Fodder Crops and Grass in UAA	%	75	48	22
Stocking density	LU[a] ha^{-1} FCG	1.5	1.5	1.2
Inputs				
Concentrate feed use	kg cow^{-1} or goat^{-1} yr^{-1}	804	1703	546
N input mineral fertiliser	kg ha^{-1} UAA yr^{-1}	60	90	74
N input organic fertiliser	kg ha^{-1} UAA yr^{-1}	27	10	12
N input concentrated feed	kg ha^{-1} UAA yr^{-1}	32	38	61
N input symbiotic fixation	kg ha^{-1} UAA yr^{-1}	32	4	7
Total N input	kg ha^{-1} UAA yr^{-1}	151	142	154
Diesel use	kg ha^{-1} UAA yr^{-1}	105	111	88
Electricity use	kWh ha^{-1} UAA yr^{-1}	344	262	264
Output				
Milk production	kg FPCM[b] cow^{-1} or goat^{-1} yr^{-1}	7758	8676	756
Milk fat content	%	4.3	4.1	3.8
Milk protein content	%	3.4	3.3	3.2
Milk-sales portion of total sales[c]	%	71	70	76
Surplus of N farm-gate balance	kg ha^{-1} UAA yr^{-1}	90	86	88

[a] "Livestock Unit," defined according to the French system (OJFR, 2000)

[b] FPCM is fat and protein corrected milk, i.e. 0.337 + 0.116 × %fat + 0.06 × %protein × kg milk sold (Thomassen and de Boer, 2005)

[c] Used for LCA economic allocation

Dairy farms examined in this study differed with respect to mean values for farm structure, input use and output level (Tab. 1). Relative to Cow B farms, Cow T and Goat T farms had a larger usable agricultural area (59 vs. 161 and 120 ha), with a lower percentage used for fodder crops and grass (75 vs. 48 and 22%). Relative to Cow B, livestock density for Cow T was similar, for Goat T it was lower

(1.5 vs. 1.5 and 1.2 LU ha^{-1}). Use of concentrated feed per cow or goat was higher for Cow T and lower for Goat T farms than for Cow B farms (1703 and 546 vs 804 kg cow^{-1} or $goat^{-1}$ yr^{-1}), total N input was similar, diesel use was similar for Cow B and Cow T, but lower for Goat T (105 and 111 vs 80 kg ha^{-1}) and electricity use was higher for Cow B than for Cow T and Goat T (344 vs 262 and 264 kWh ha^{-1}). Mean annual FPCM production per cow or goat was 7758 kg for Cow B, 8676 for Cow T, and 756 for Goat T. The proportion of milk sales in total farm sales was similar for Cow B and Cow T, and higher for Goat T (71 and 70 vs. 76%). However, this proportion was more variable for Cow T than for Cow B (data not shown). Surplus of the N farm gate balance (N inputs – N outputs) was similar for the three farm types.

Impacts of cow and goat dairy farms

Our results for cow T milk at the farm gate are within the range of impact values summarised by Thomassen *et al*. (2008) for all impacts except AC, where cow T presented lower values. When expressed per 1000 kg of FPCM and relative to Cow B, impacts for Cow T were 12 –70% higher, and impacts for Goat T were 145 – 263% higher. When expressed per ha of land occupied Cow T farms were similar to Cow B farms for EU and NE, while AC, CC and TT were lower for Cow T farms than for Cow B farms (Tab. 2). Per ha of land occupied, impacts for Goat T farms were similar to Cow B farms for EU, AC and TT, CC was lower for Goat T farms than for Cow B farms, but NE was higher (Tab. 2).

Tab. 2: Mean impacts (1) per 1000 kg fat and protein corrected milk (FPCM) and (2) per ha of land occupied for dairy cow farms in Bretagne (Cow B, n = 46), for dairy cow farms in Pays Thouarsais (Cow T, n = 13) and for dairy goat farms in Pays Thouarsais (Goat T, n = 6).

		Per 1000 kg FCPM			Per ha of land occupied		
Potential impact	Units	Cow B	Cow T	Goat T	Cow B	Cow T	Goat T
Eutrophication	kg-eq. PO_4	6.2	9.3	14.3	40.1	39.0	39.6
Acidification	kg-eq. SO_2	7.3	8.2	16.1	48.2	34.5	47.4
Climate change (100 yr)	kg-eq. CO_2	880	1033	1272	5806	4347	3700
Terrestrial toxicity	kg-eq. 1.4-DCB	1.6	2.0	3.8	10.5	8.2	10.2
Non-ren. energy use	GJ	3.0	5.1	7.9	19.6	21.0	22.5
Land occupation	m^2 yr^{-1}	1530	2431	3481			

Presentation of these results (Tab. 2) in a meeting with project partners led to vivid discussions concerning the use of economic allocation to allocate the farm's impacts to milk, animals and crop products. It was argued that this choice might introduce artefacts, the more so on farms in which cash crops contribute to a major extent to total farm sales. This argument makes sense, because although Cow B farms and Cow T farms are similar with respect to the proportion of milk in farm sales, Fodder Crops and Grass in UAA is much lower for Cow T than for Cow B (48% vs. 75%, Tab. 1), indicating that crop production was more important for Cow T farms than for Cow B farms.

We therefore examined a subset of five Cow T farms differing strongly with respect to the proportion of milk sales in the total farm sales (34-86%). For these farms cash crops were excluded from the system. This involved excluding inputs (farm land, pesticides, fertilisers, diesel, tractors and other machines) used for these crops as well as excluding sold crop products from the calculations in EDEN. Input use for cash crops could partly be based on available data (e.g. fertiliser rates for each crop), however for some inputs (e.g. diesel use) specific data for each crop were not available, and we used estimations based on our (unpublished) references.

These preliminary results show that, when cash crops are included in the system, impacts per 1000 kg of milk increase with the % of milk sales in total sales, whereas this is not the case when cash crops are excluded from the system (Fig. 1 and 2).

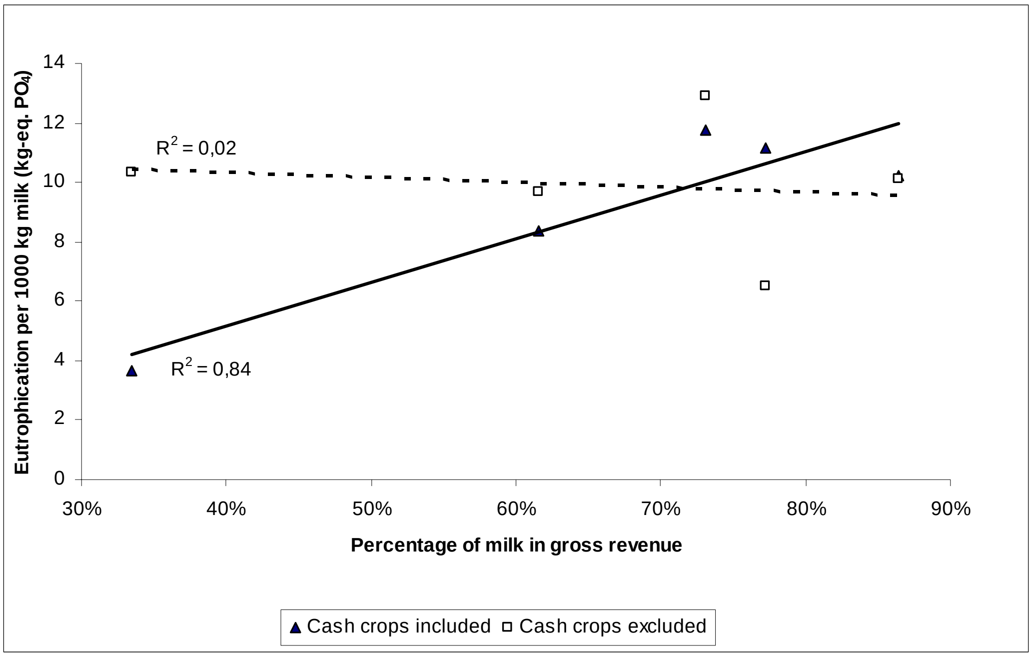

Fig. 1: Eutrophication per 1000 kg of FPCM for five dairy cow farms, as a function of the percentage of milk sales in total farm sales for the farm including its cash crops (triangles) and for the farm excluding its cash crops (squares).

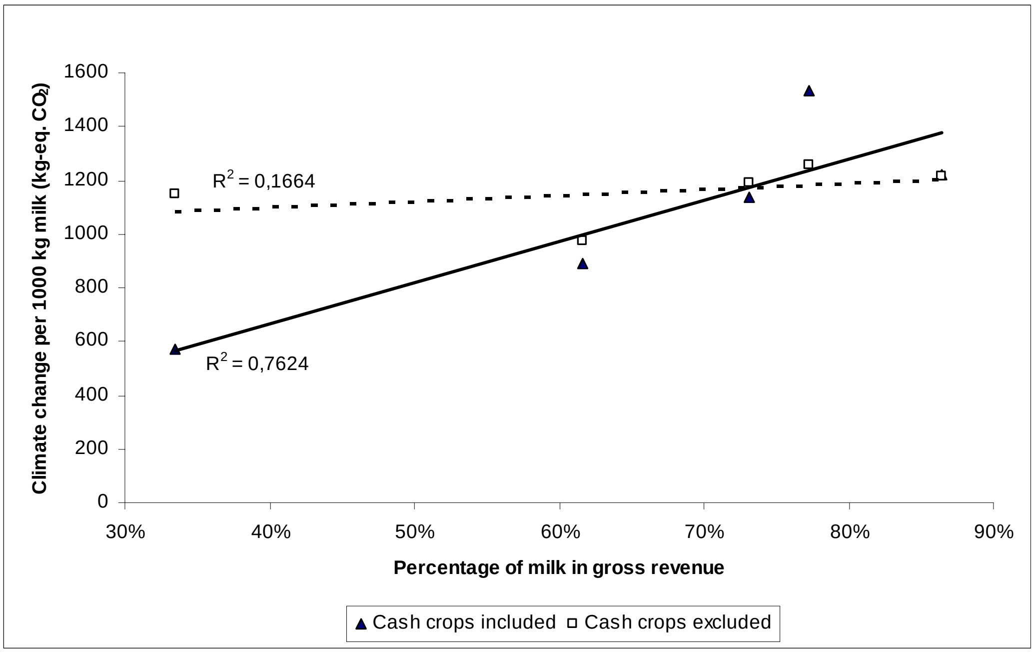

Fig. 2: Climate change per 1000 kg of FPCM for five dairy cow farms, as a function of the percentage of milk sales in total farm sales for the farm including its cash crops (triangles) and for the farm excluding its cash crops (squares).

Impacts of a cow dairy chain

We divided the cow milk chain originating in the Pays Thouarsais in four stages: 1) production and delivery of farm inputs, 2) farm operation, 3) transport to and operation of the dairy plant and 4) post-plant transport of products (Tab. 3). The contribution of impacts associated with farm inputs to total inputs for the milk chain is variable, ranging from 9% for EU to 49% for NE. For all impacts except NE (to which it contributes 15%) farm operation is the dominant stage of the milk chain, its contribution ranges from 65% (CC and TT) to 86% (EU). The actual processing of milk, including milk collection at the farms, varies from 0% (LO) to 8% (CC) and it thus is the stage that contributes least to the impacts evaluated here. The transport of products from the plant to retailers contributes to a variable extent to overall impacts: 0% (LO) to 28% (NE). These results are in accordance with the findings of Foster *et al*. (2006), who found that production up to the farm gate contributed most to all impacts considered, but least so for energy use.

The search for options to reduce impacts associated with this milk production chain should neglect none of the four stages. To explore the potential contribution of improvements for the farm operation stage to overall impacts of the milk chain we identified, within the set of thirteen dairy cow farms, the two farms with the lowest values for NE per 1000 kg FPCM and the two farms with the highest values for this impact. Based on the average of each set of two farms we then calculated characteristics for “low energy milk” and “high energy milk” and assessed impacts of milk chains based on these types of milk (Tab. 3). Results for low energy milk reveal that improvements at the farm stage may have a large potential to reduce impacts of the chain. This holds for NE (-20%), but also for the other impacts, with reductions ranging from 24 – 39%. Conversely, the use of high energy milk leads to increased impacts relative to the average milk scenario. It remains obviously to be seen at what cost farms with high values of NE per 1000 kg of FPCM can reduce their energy use.

Tab. 3: Impacts associated with the production of 1000 kg FPCM, its transformation and the transport of products produced. Contribution (in %) of farm inputs, farm operation, transport to and operation of dairy plant and post-plant transport of products to total impacts. Average milk is based on data for thirteen farms. Low and high energy milk give total impacts using data for two sets of two farms within the thirteen-farm sample with lowest and highest values for NE per 1000 kg FPCM.

		Average milk					Low energy milk	High energy milk
Potential impact	Units	Inputs	Farm	Dairy	Transport	Total	Total	Total
Eutrophication	kg-eq. PO_4	9%	86%	4%	1%	9.8	6.0	11.6
Acidification	kg-eq. SO_2	19%	73%	1%	7%	8.9	6.3	11.4
Climate change (100 yr)	kg-eq. CO_2	23%	65%	2%	10%	1167	890	1431
Terrestrial toxicity	kg-eq. 1.4-DCB	22%	65%	4%	9%	2.3	2.9	6.9
Non-ren. energy use	GJ	49%	15%	8%	28%	7.9	6.3	10.3
Land occupation	m^2 yr^{-1}	15%	85%	0%	0%	2436	1733	3327

Discussion and conclusions

This paper presents the first results of the PaRMEELI project, which aims to analyse the environmental impacts of regional dairy chains in order to identify improvement options across the entire chain. Per 1000 kg of FPCM produced, Thouarsais cow milk had somewhat higher impacts than Bretagne cow milk, but per ha of land occupied its impacts were similar to or less than those of cow milk from Bretagne. However, and probably more importantly, results of a preliminary analysis on the effect of the exclusion of cash crops from the system suggest that the use of economic data to allocate impacts to milk, animals and cash crops might not be the best option for these farms, some of which make more money from crops than from milk.

Per 1000 kg of FPCM produced, Thouarsais goat milk had higher impacts than Thouarsais cow milk, but per ha of land occupied its impacts were not very different from those of Thouarsais cow milk.

However, given the fact that the percentage of the farm agricultural area used for fodder crops and grass is particularly low (22%) on these farms, these results should be re-examined by exploring the effect of the exclusion of cash crops from the system as for the Cow T farms.

For all impacts except NE, farm operation was the stage contributing most to overall impacts of the cow milk chain examined here, impacts of farm inputs came second. For the post-farm dairy chain, impacts associated with the transport of products to retailers were more important that those of the dairy plant itself. Non-renewable energy use presented a strikingly different pattern of contribution: farm inputs contribute most, transport to retailers came second, with farm operation contributing much less than for other impacts, while the dairy plant contributed least, but more than the other impacts.

These results will be analysed with project partners in order to identify the most promising improvement options for all stages of the milk chain.

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Life cycle assessment of a pilot plant for the must enrichment by reverse osmosis

B. Notarnicola[1], G. Tassielli[1], P. Renzulli[2], E. Settanni[1]
[1]Department of Commodity Science, University of Bari, via C. Rosalba n. 53, 70124 Bari, Italy, b.notarnicola@dgm.uniba.it
[2]Renzulli Consulting Engineering, Bari, Italy

Keywords: LCA, must enrichment, reverse osmosis, wine

Abstract

The enrichment of wine musts via the method of concentration at high temperature is currently widespread. It requires large quantities of energy and it can affect in a negative way the organoleptic characteristics of the musts. Reverse osmosis concentration methods, on the other hand, are still poorly investigated in the wine making process even if they offer promising development prospects, mostly with regard to the use of semi permeable membranes. Itest, a southern italian mechanical company, specialised in the building and set up of wineries, has developed a pilot plant for the study of the characteristics of the membranes to be used with native vines. In this paper, which is extracted from a research project financed by the Regione Puglia, the Life Cycle Assessment of the reverse osmosis plant for the must enrichment is performed. In this study the use of Life Cycle Assessment allowed not only to achieve an environmental improvement, but also to identify an optimized machinery set-up.

Introduction

The alcohol enrichment of wine must is one of the most common practices in the wine making process. Today the method of concentration at high temperature is the most frequently used one. It requires great energy quantities and it can affect in a negative way the organoleptic characteristics of the product. On the contrary, reverse osmosis (RO) concentration methods are still poorly investigated in the wine making process, mostly with regard to the semi permeable membranes to be used (Baker, 2004) and various aspects such as energy, materials and environment of the RO systems, also considering the process variables. Moreover these methods offer promising development prospects since the new OCM wine, approved in December 2008, proposes to progressively eliminate the financial support given to wine concentration achieved via the addition of rectified concentrated must.

The University of Bari and Itest, a southern italian mechanical-hydraulic company specialised in the building and set up of wineries, which has developed a reverse osmosis pilot plant for the must enrichment, have carried out a research project financed by the Regione Puglia, whose general goal is to acquire the knowledge necessary for assessing the environmental characteristics of the reverse osmosis plants for the enrichment of musts, which in turn is essential for the optimisation of the development of such plants. It has been agreed that the most suitable application methodology for this study is the Life Cycle Assessment of the product (LCA), standardized by the rules of the series ISO 14040 that can provide useful indications in the case of "Design for Environment" (Notarnicola *et al*., 2003).

Method / Approach

Goal and scope definition

The objective of this study is to carry out an investigation on the environmental performance of the must concentration process using semi-permeable membranes. The "functional unit" of the study is 1000 L of must for which the aimed enrichment is 1 alcoholometric degree from 10° to 11°.

The system studied involves a reverse osmosis plant that performs must enrichment, carried out directly in the cellar; the examined system includes the extraction and the processing of raw materials, the production of the plant, transport and distribution, use, re-use and maintenance, recycling of the components and final disposal (Notarnicola *et al*., 2001). In particular, the phases that have been considered are:

- The production of the RO plant and all of its components starting from the raw materials necessary for the production;
- the transport of the plant to the cellar;
- the actual concentration of the must;
- the washing of the semi-permeable membranes, including the use of detergents and their production;
- the disposal of the materials of the plant via recycling at the end of its useful life;
- the final disposal of the residues at the end of the useful life of the RO plant,

The collected data for each the above mentioned process unit regard the energy consumption and air, water and soil emissions. The time span for the data is 10 years; in particular, the data regarding the must concentration refers to the last two years. The rest of the data, collected on site and via literature, refers to the last ten years (Asano *et al*., 2007; Baker, 2004; AWWA, 1996; MWH, 2005; WEF, 2005; Tchobanoglous *et al*., 2002).

For this study the Italian electrical mix has been considered; for imports, a specific electrical mix from the country of origin has been considered. A useful lifetime of 20 years is assumed for the enrichment plant together with a work capacity of 2000 hours/year. The final disposal considers a recovery of 90% of the weight of the components that are recycled, whilst the remaining 10% is simply disposed. The lifetime of the membranes varies from 1000 to 2500 hours of use, depending of the type of use and the maintenance performed; as a precautionary measure a lifetime of 1000 hours is assumed. The plant can work up to 48 hours nonstop before having to perform membrane cleaning; for this study a work cycle of 20 hours together with a 4 hour cleaning cycle is assumed. Also, in order to determine the transportation aspects, a distance of 100 km between the RO plant production company and the cellar is assumed.

The impact assessment methodology which has been used is the “problem-oriented” one (Heijungs, 1992), through phases of classification and characterisation. The following impact categories have been considered: primary energy consumption (EC), abiotic resource depletion (ADP), global warming (GWP), acidification (AP), photochemical oxidant creation (POCP), human toxicity (HTP), fresh aquatic eco-toxicity (FAETP), terrestrial eco-toxicity (TETP), nutrification (NP). The characterisation factors for all the impact categories, with the exception of the EC, are taken from the CML 2000 Guide (Guinée, 2000). For the EC category the values of primary energy are taken into account.

Inventory

The creation of an inventory of the RO must enrichment plant considers the analysis of the production cycle of the system itself.

The production cycle of the system is typical of metalworking industries. The primary material used is steel. The structural shaped steel that arrives at the production site is cut via band saw to obtain the desired shapes. Some of these components are drilled to prepare them for the next phase.

The assembly phase involves soldering and joining, via bolts, components produced on site or purchased (membranes, monometers, pneumatic pressure switches etc.). The final system is made up of a structural part, an electrical system and a hydraulic one (Notarnicola *et al*., 2007).

Tab. 1: summarises the energy consumption, auxiliary material and products used in the production phase of the reverse osmosis plant together with the output of this production. As can be noticed there are no particular impacts in this phase since the production is simple and only involves cutting and assembly operations. The largest effort on behalf of the firm relates to the research, design and development of this system.

Tab. 2 shows the typical materials used subdivided for each main component. The mass of the system is approximately 294,5 kg; steel is the main material used, followed by aluminium, copper and polyamide resins that make up the membranes (Scott *et al.*, 1996; Winston *et al.*, 1992; Peinemann *et al.*, 2008; AWWA, 2007).

Tab. 1: Consumption of electric energy, materials and auxiliary products of the production phase.

Inputs	**Units**	**Quantities**
Primary components	kg	299.5
Cutting fluid	g	300
Welding rods (Aisi 304 steel)	g	500
Abrasive discs (phenol resin/aluminium oxide: 85%/15%)	g	2000
Argon gas for welding	m^3	18
Band saw blade (steel)	g	300
Drill bits HSS (steel)	g	100
Pickling paste	g	500
Electrical energy	kWh	58
Outputs		
Concentration Plant	kg	294.5
Scrap left over Aisi 304 Steel	kg	5.0
Unused leftovers	kg	3.2

Tab. 2: Quantity of materials that make up the system.

Materials	**Weight (g)**	**Materials**	**Weight (g)**
Steel Aisi 304	193654	Polyester	720
Steel Aisi 316	9740	ABS	10240
Steel C40	33000	Polyurethane	1844
Iron	2075	Brass	222
Cast Iron	1955	Copper	16790
Rubber NBR	761	Aluminium alloy	710
Siliconic rubber	641	Aluminium	16303
EPDM for foods	50	Lubrication oil	300
Nylon	591	Ceramic	441
Polyamide	2160	Glycerine	12
PVC	1965	Glass	20
Plexiglas-PMMA	293	**Total**	**294487**

The most relevant phase, with regards to this study, is the one during which the actual enrichment of the musts is performed via the semi-permeable membranes.

Factors that influence the performance of the system in terms of permeate produced per hour are the temperature of the surroundings, the temperature of the must, the initial and final alcohol grade, the kind of winemaking (red or white), the amount of suspended solids in the must and the kind of water used to wash the system before re-use (Schaefer *et al.*, 2003; Mulder, 1996; Munir, 1998).

All these factors can lead to operational conditions that vary from a minimum of 50-60 L/h of permeate removed from the must to a maximum of 250-300L/h in the case of products that have been extremely clarified after fermentation.

During use the system consumes 7.6 kWh of electrical energy per hour and other materials such as lubricating oil.

For the RO system to work properly a series of washing and maintenance cycles are needed; in fact to maintain a decent permeate output the membranes have to be frequently washed with water and citric acid or potassium carbonate. Other less frequent maintenance tasks include the substitution of the mechanical filter, of the oil in the piston driven pump and of the filters every 1000 hours of use of the machine.

Maintenance tasks include the substitution of the mechanical filter, of the oil in the piston driven pump and the substitution of the membranes which for this study is assumed to happen every 1000 hours of use of the machine.

The useful lifetime of the machine can be estimated in 30,000-50,000 hours of work; as mentioned above for this study the lifetime is assumed to be 40,000 hours.

In order to enrich the functional unit of must, i.e. obtain a must from which it would be possible to obtain a wine with a higher alcohol content by one degree (from 10°to 11°), it is necessary to remove from the must 91 L of water. Since the method involved is a subtractive one, the final quantity of enriched must is 909 L, therefore inferior when compared to other additive methods. Furthermore, by analysing the collected experimental data, an output of 75 L/hour of permeate is considered.

In order for the plant to achieve the desired experimental enrichment of the functional unit, the system has to work for 1.2 hours.

Results

Fig. 1 shows the characterization of the system per phases. From this it is possible to notice that for most of the examined impact categories the most impacting one is the use of the RO plant due to the energy consumption of that phase followed by the membrane washing phase.

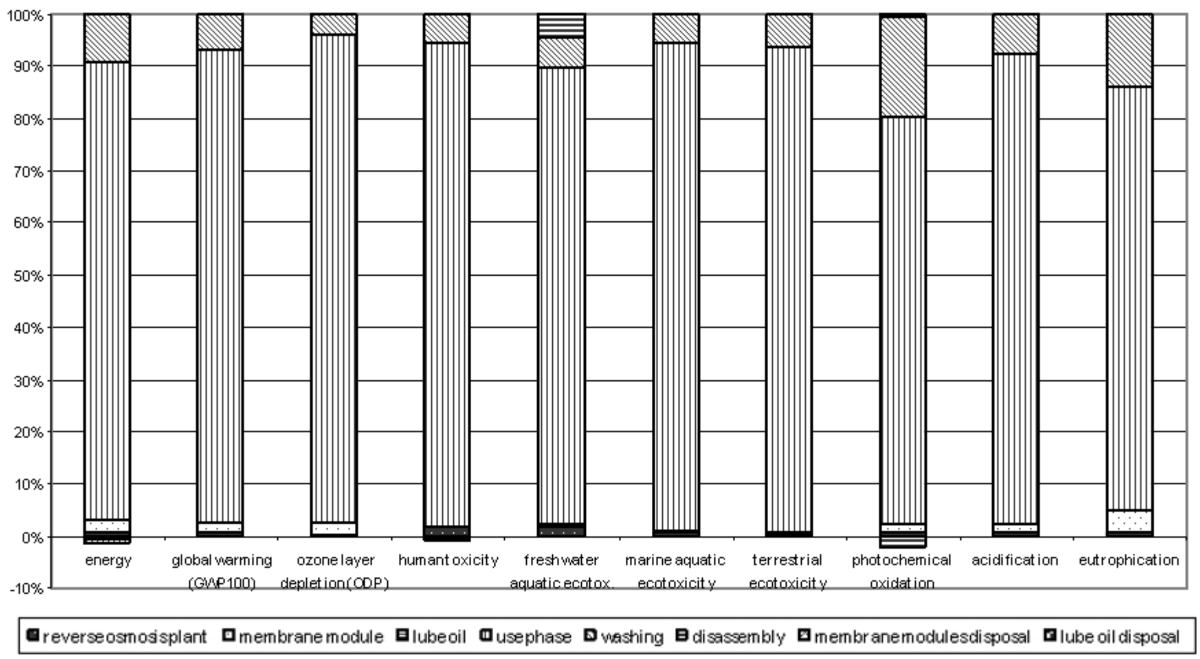

Fig. 1: Characterization of the system per phases

In this system the energy consumption in the disposal phase of the components can be neglected; the energy consumption incorporated in the materials that make up the plant and the energy consumption incorporated in the recycled materials are also fairly modest. This result is coherent with those presented in other studies on the environmental impact of the machinery since the energy used, or environmental impact associated with the obtainment of the machine, must be divided by a large number of manufacturing cycles, hence producing modest results.

The phases with the largest consumption are those related to the actual use of the RO plant; in particular the enrichment phase and the membrane washing phase contribute respectively to the 92.7% and 7.2% of the overall energy consumption of the analysed system. The most widely used source of energy is electrical and as a consequence the direct energy consumption is inferior to the indirect one with respectively 37.3% and 63.7% of the total energy consumption.

Discussion

Identification of the best opportunities to improve the system and experimental check

The research focused on finding solutions to decrease environmental impacts through the identification of the best opportunities to optimize the system under study via an improvement assessment. Therefore tests were carried out with the specific goal to identify the machine setup that would allow the best trade off between efficiency and environmental impact during must enrichment.

Aspects that were expected to be dealt with during this phase of the study were material selection, the reduction of the impact of production, energetic efficiency improvement during the RO plant use, design for recycling an re-use, extension of the lifetime of the plant and its components and the design for the end of the life of the plant.

In reality, the inventory results and the evaluation of the impacts showed that the most critical phase was the actual utilization of the RO plant. This phase in effect weights the most on the eco-indicator for both of its components: operation and washing. The other phases impact less; in fact, the system already presents a large quantity of components that are recyclable; the assembly and disassembly of the product are already quite simplified and cannot be considered as having a significant environmental impact.

The research therefore concentrated on the identification of solutions that would reduce the impact during the utilization phase. Hence a quantitative and qualitative evaluation was carried out regarding the possible advantages obtainable in terms of energy balance, environmental impact and economic profile of the proposed solutions and regarding the feasibility and economic convenience of the adopted innovations.

The impact during the phase of utilization mainly depends on the following factors:

- duration of the concentration stage;
- efficiency of the concentration stage;
- productivity in terms of permeate produced;
- amount of membrane fouling.

These factors contribute to the total direct electrical energy consumption and also contribute to the number of times the membranes have to be washed which, in turn, determines consumption of energy and auxiliary washing material and the disposal of the waste water.

As a consequence the research focused on trying to find solutions that could affect the above mentioned factors. In particular, the study concentrated on identifying possible parameters that had not yet been considered, that could affect the operation of the RO plant during enrichment.

Previous studies showed that the enrichment obtained was proportional to the pressure exerted on the must and membranes. Higher working pressures imply larger quantities of produced permeate; also the concentration, in terms of reducing sugars contained in the enriched must, increase proportionally. All the previous studies and experimentations carried out considered only this parameter. Also all current literature emphasizes the pressure as the driving parameter.

However the performance of the RO plant is affected by controllable parameters and uncontrollable ones. Among the latter are the ambient temperature, the must temperature, the initial alcohol content of the must, the type of must (white or red) and the type of water used for the membrane cleaning. It is assumed that these factors affect all the concentration process in the same manner.

This study identified another control parameter that had not been previously considered, the velocity of the must flux entering the RO plant. Previously this parameter had not been judged as an important element influencing the concentration process. Moreover, in the past, it was assumed that the maximum possible must flux entering the machine would only bring about a better membrane cleaning during the concentration operation. As a consequence the must flux entering the machine was always kept at a maximum of 55 L per minute.

In this study, it was assumed that this factor could greatly influence the concentration process. Hence the RO plant was modified by installing an electrical inverter that could modulate the power and hence speed of the high pressure pump.

New tests were then carried out on using the modified RO plant. The methodology used for the performance of the experiments is the Design of Experiments (DOE) (Anderson & Whitcomb, 2007). The results of the led experiments show that, unlike what was deemed by many experts in the field, the processing with the reduced incoming must flow consumed less electric power, it didn’t reduce the quantity of the permeate produced, rather it increased. Further the chemical analysis led on the enriched musts and on permeate show a greater efficiency of the concentration operation. The gained advantages of using a plant that has half the maximum incoming must flux (27.5 L/h) were:

- less energy consumption: 6.17kWh/hour as opposed to 7.6 kWh/h;
- better performance in terms permeate produced: 113 L/h as opposed to 106.5L/h;
- the obtained must is more concentrated: sugar content increased from 217.7g/L to 228.5g/L
- the waste waters from the membrane cleaning were not as fouled: COD values decreased from 18000 mg/L to 16000 mg/L;

Having identified these improvements a comparative impact assessment was performed between the original RO plant and the new one (Fig. 2)

The adopted innovations brought about an improvement, in the environmental profile of the system that ranges, in the single environmental impact categories, from a minimum of 25.3 to a maximum of 26.6%.

In compliance with the objectives of this study, the sugar content of musts has been chosen as the only relevant quality parameter for musts, and the functional unit has been defined accordingly. After the improvement of the machinery set-up, the LCA has been carried out while keeping the same functional unit as before. The alcoholometric degree of musts being constant, several quality parameters, such as polyphenols and tannins contents, may have changed, however. Therefore, if musts’ quality parameters other than sugars had been considered, the functional unit would have been defined differently to represent a multi-functional system.

Moreover an economic evaluation was performed on the adopted innovations that allowed the following savings:

- electric energy and manpower in the concentration phase;
- electric energy, products and manpower in the membrane washing phase.

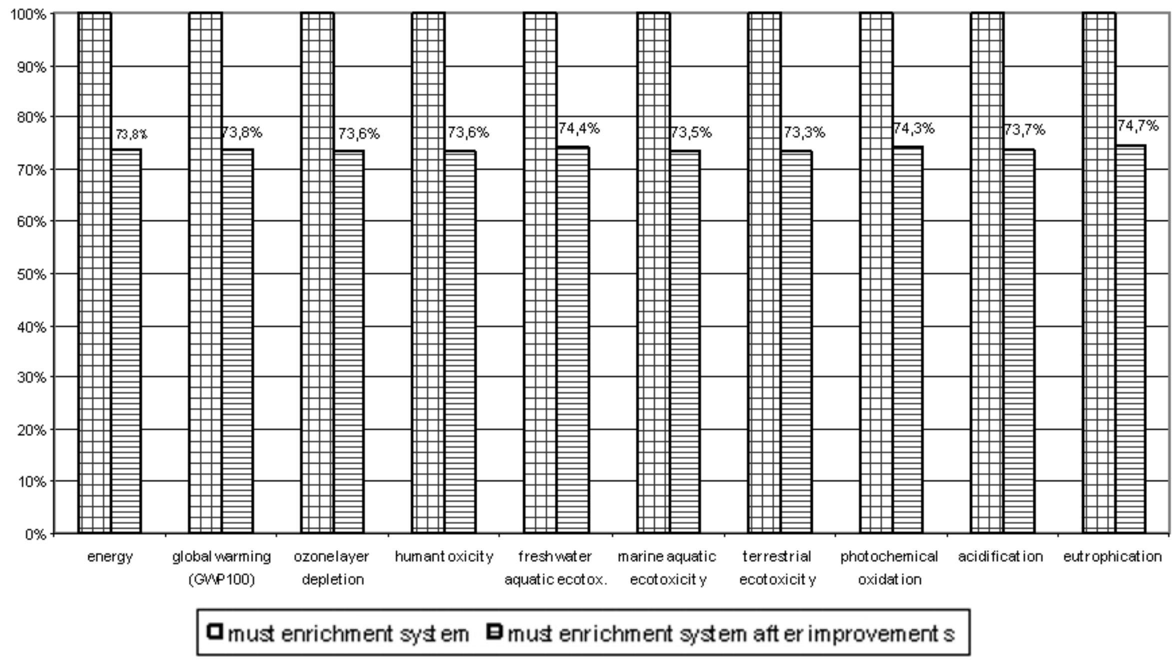

Fig. 2: Characterization of the system before and after the improvements

Conclusion

The inventory and impact assessment results show that the utilization phase, which could be divided in two subphases, enrichment process and cleaning, absorbs most of the direct energy and material consumption and that the preproduction, the production and the final disposal phases have a minor impact which is typical for most of the studies on the life cycle of machines. The research focused on finding solutions to decrease environmental impacts through the identification of the best opportunities to optimize the system under study, going through an improvement assessment. Therefore tests were carried out with the specific goal to identify the machine setup that permitted the best trade off between efficiency and environmental impact during the enrichment of musts. The methodology used for the performance of the experiments is the Design of Experiments (DOE) (Anderson & Whitcomb, 2007).

The results of the experiments show that, unlike what was deemed by many experts in the field, the enrichment with a reduced incoming must flow consumed less electric power and did not reduce the quantity of permeate produced but rather it increased it. Furthermore the chemical analysis performed on enriched musts and on permeate show a greater efficiency of the concentration process. The results of the research have overall improved the RO plant under study and have increased its added value. The research has increased the environmental knowledge in the application field of reverse osmosis technology with semi permeable membranes in the wine sector.

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Importance of human excretion in LCA of food. Case study of the average Spanish diet

Ivan Muñoz[1], Llorenç Milà i Canals[2], Amadeo R. Fernández-Alba[1]
[1] Department of Hydrogeology and Analytical Chemistry, University of Almeria, 04120 Almeria, Spain. E-mail: ivanmuno@ual.es
[2] Centre for Environmental Strategy, University of Surrey, Guildford, Surrey, GU2 7XH, UK.

Keywords: Life Cycle Assessment, food, human excretion, wastewater treatment, nutrient cycle, carbon cycle

Abstract

The aim of this work is to find out to what extent human excretion, a life cycle phase often excluded from most food LCA studies, is relevant in the context of a person's overall food intake. A case study has been carried out which deals with the average Spanish diet, including the entire life cycle of food (with the exception of packaging materials): agricultural and animal production, industrial processing, distribution and retail, home storage and cooking, solid waste management, and human excretion. In order to include all the processes related to human excretion such as metabolism, toilet use, and wastewater treatment, a recently developed model has been used. Three impact categories are assessed; namely - Global Warming Potential, Acidification Potential, and Eutrophication Potential, along with Primary Energy Use as an environmental indicator. The results show that although food production clearly appears as the main hotspot in the Spanish diet, human excretion, as well as further wastewater treatment, is by no means a negligible process, especially in terms of Eutrophication and Global Warming, where it is the second most important source of emissions. On the other hand, its contribution to Acidification is almost negligible, and it is rather low in Primary Energy Use.

Introduction

When a life cycle perspective is taken into account, food consumption stands out as one of the most resource-demanding and polluting daily activities in our lives. Several studies, at both the national and international level, have identified food supply as one of the main contributors to environmental impacts caused by private consumers (Nijdam *et al.*, 2005; Tukker *et al.*, 2006). This explains why LCA has focused on food since the early 1990s (Weidema, 1993), being a suitable tool for finding ways to make food production and consumption patterns sustainable.

Although research has mostly addressed individual food products, or particular stages of their life cycle (farming; processing; packaging; transport; home processing and storage, as well as waste management), some authors have also studied the environmental impacts of food supply from a dietary perspective, either to identify the food items that are the most polluting, or to compare dietary choices (Carlsson-Kanyama, 1998; Kramer *et al.*, 1999; Jungbluth *et al.*, 2000; Carlsson-Kanyama *et al.*, 2003; Heller and Keoleian, 2003).

Human excretion remains the least studied life cycle stage at both the product and dietary levels. So far, only the fate of nutrients has been modelled in some LCA studies. Sonesson *et al.* (2004) proposed a systematic procedure to include emissions of chemical oxygen demand (COD), as well as nitrogen and phosphorus resulting from food composition. However, human excretion as a whole was not addressed. These processes are usually excluded from the system boundaries although the biochemical transformation of food in the human body leads to pollutants being released into the air and into the water. These should be quantified and assessed in a similar way to organic matter when it is treated in a landfill or a composting plant.

Recently, a simple model to include human excretion in LCA of food products has been developed (Muñoz *et al.*, 2008). This model uses the basic nutritional composition of food items to calculate the emissions of several pollutants into the air and into wastewater, as well as the consumption of auxiliary materials and energy which is related to toilet use. Emissions into wastewater, when used as the input to a wastewater treatment model like that of Doka (2003), allow the LCA practitioner to fully complete the balance of materials and energy concerned with food supply if the results from these models are integrated with data from previous life cycle stages. This is done in Muñoz *et al*. (2008), in a case study on a single product, namely broccoli, where the importance of the human excretion stage is compared to those of production, distribution, and home processing. The aim of this work is to carry out a similar assessment, but at the dietary level. This is done by evaluating all the food taken in by a person over a one year period when consuming the average Spanish diet.

Method

The average Spanish diet

Detailed data on food purchases by the Spanish population is available through official statistics (Ministerio de Agricultura, Pesca y Alimentación, 2006a). Tab. 1 shows the composition of food purchases by weight in 2005, including both purchases for household and outside household consumption (restaurants, and institutions such as schools and hospitals).

Tab. 1: Per capita food purchases (net weight) in Spain in 2005.

Product group	**kg**	**%**
Eggs	14	1.6
Meat products	66	7.5
Fish and seafood	37	4.2
Dairy products	143	16.2
Bakery products	70	8.0
Vegetable oils and fats	22	2.5
Vegetables	108	12.2
Fruit	103	11.7
Beverages	174	19.8
Bottled water	68	7.7
Other	77	8.7
Total	881	100.0

Source: Ministerio de Agricultura, Pesca y Alimentación, 2006a.

Scope of the study

From the product groups in Tab. 1, we have only excluded bottled water (7.7% of total purchases), and those for which inventory data related to production has not been obtainable. This applies to some items included in the 'other' category: ready meals, honey, soups, sauces, nuts, and products unspecified in weight, which altogether account for 3% of the total purchases in Tab. 1. In the present study, we assess the life cycle impact of 787 kg of food per person per year.

The study includes all the upstream and downstream operations required for the consumer to have ready-to-eat food: farming, industrial processing, distribution and retail, home storage and cooking, human excretion, as well as wastewater treatment. The only issue excluded for all the food items studied is production of packaging, due to the amount of effort that would be required to collect inventory data for such a diverse group of products.

The functional unit chosen is the supply of food for a Spanish citizen in the year 2005, which as stated above amounts to 787 kg of food.

Life Cycle Impact Assessment (LCIA) has been focused on a limited set of indicators. The following impact categories are included: Global Warming Potential (GWP); Eutrophication Potential (EP); and Acidification Potential (AP), applied at the characterisation level using the characterisation models of the CML 2000 Method (Guinée *et al.*, 2002). Primary Energy use (PEU) has also been used as an environmental indicator, measured in MJ.

Food production

Due to the substantial amount of products that need to be included in the assessment, it has been based on already published background inventory data. Although we have managed to include 90% of the food weight present in Tab. 1, this does not mean that specific inventory data were found for each product group included. In fact, gaps were encountered for many products, as can be seen in Tab. 2. At present, the only LCA database including the basic food production processes in Europe is the LCA Food database (Nielsen *et al*., 2003) developed in Denmark, which has been used as the basic source of information for this study.

Many missing products in the Food LCA database have been modelled using data from Carlsson-Kanyama and Faist (2000). The latter includes mainly inputs from the technosphere associated to agriculture and industrial processing, such as fuels and electricity, fertilizers, pesticides, etc... In order to obtain product datasets from these data, background inventory data from the LCA food database have been used for these inputs. Fertilizer-derived emissions (ammonia, dinitrogen monoxide, N and P leaching to groundwater) have been estimated for these missing products following the recommendations by Milà i Canals *et al*. (2007).

In all datasets the Spanish electricity production profile has been used, as included in the Ecoinvent database (Dones *et al*., 2003). Another change made to original LCA Food datasets is to exclude artificial heating of greenhouses, as this is more representative of Spanish agriculture.

Tab. 2: Data sources used for food production modelling.

Products	**Modelled as**
Eggs	Eggs from LCA Food database
Fresh meat: chicken, ovine and goat, rabbit, and unspecified fresh meat	Fresh chicken from LCA Food database
Fresh pork meat	Fresh pork tenderloin from LCA Food database
Fresh bovine meat	Fresh beef fillet from LCA Food database
Frozen meat	Frozen chicken meat from LCA Food database
Processed meat: salted meat, lard, cured ham, knackwurst sausages, other processed meat	Fresh ham from LCA Food database
Fresh cod and tuna	Fresh cod from LCA Food database
Sole	Fresh flatfish from LCA Food database
Hake, sardine, salmon, and unspecified fresh fish	Fresh mackerel from LCA Food database
Trout	Fresh trout from LCA Food database
Frozen sole	Frozen flatfish fillet from LCA Food database
Frozen cod	Frozen cod fillet from LCA Food Database
Frozen hake, salmon, and unspecified fish	Frozen mackerel fillet from LCA Food Database
Fresh seafood	50% fresh mussels and 50% fresh shrimps from LCA Food Database
Frozen seafood	Frozen shrimps from LCA Food Database
Canned fish	Canned tuna from Hospido *et al*. (2006)
Full milk and milk shakes	Full milk from LCA Food database
Yoghurt	Yoghurt data from Carlsson-Kanyama and Faist (2000)
Semi-skimmed milk	Low fat milk from LCA Food database
Skimmed milk	Skimmed milk from LCA Food database
Cheese and unspecified dairy products	Cheese from LCA Food database
Bread	Bread from LCA Food database
Biscuits and cakes	Rolls from LCA Food database
Chocolate	Chocolate data from Carlsson-Kanyama and Faist (2000). Includes only industrial processing energy.

Products	Modelled as
Coffee	Coffee data from Carlsson-Kanyama and Faist (2000). Includes only industrial processing energy.
Spaghettti	Spaghetti data from Carlsson-Kanyama and Boström-Carlsson (2001)
Sugar	Sugar from LCA Food database
Rice	Rice data from Carlsson-Kanyama and Faist (2000)
Legumes	Dry beans and peas from Carlsson-Kanyama and Faist (2000)
Olive oil	Rape seed oil from LCA Food database, substituting rape seed for olives as main input
Tomato and pepper	Tomato from LCA Food database
Onion and garlic	Onion from LCA Food database
Cucumber	Cucumber data from Carlsson-Kanyama and Faist (2000)
Cabbage	Cabbage data from Carlsson-Kanyama and Faist (2000)
Green beans	Green beans data from Carlsson-Kanyama and Faist (2000)
Lettuce and leafy vegetables	Lettuce data from Carlsson-Kanyama and Faist (2000)
Mushrooms, asparagus, and unspecified vegetables	Carrots from LCA Food database
Potatoes	Potatoes from LCA Food database
Frozen potatoes, processed potatoes	Potatoes from LCA Food database plus processing data from Carlsson-Kanyama and Faist (2000)
Orange, mandarin orange, lemon	Orange data from Carlsson-Kanyama and Faist (2000)
Bananas	Banana data from Carlsson-Kanyama and Faist (2000)
Apple, peach, pear, apricot, melon, watermelon, plum, kiwi, and unspecified fresh fruit	Apple data from Carlsson-Kanyama and Faist (2000)
Strawberry	Strawberry data from Carlsson-Kanyama and Faist (2000)
Grapes	Grapes data from Carlsson-Kanyama and Faist (2000)
Cherries	Cherries data from Carlsson-Kanyama and Faist (2000)
Canned fruit and vegetables	Tomato from LCA Food database plus processing data from Carlsson-Kanyama and Faist (2000)
Frozen fruit and vegetables	Tomato from LCA Food database plus processing data from Carlsson-Kanyama and Faist (2000)
Olives	Olive data from Carlsson-Kanyama and Faist (2000)
Wine	Wine data from Aranda *et al*. (2005)
Beer	Beer data from Koroneos *et al*. (2005)
Cider	Wine data from Aranda *et al*. (2005), substituting grapes for apples as main input
Juice	Data for orange and orange juice production from Carlsson-Kanyama and Faist (2000)
Soft drinks	Assumed as water with 10% sugar. Data for soft drink processing from Carlsson-Kanyama and Faist (2000)

Due to the market-based approach used in the LCA Food database, its models only include processes influenced by marginal changes in demand. Thus, in the case of milk production, which is determined by quotas from the European Union, no burdens are allocated to its agricultural stage. Since this LCA case study is attributional (for a discussion on attributional/consequential LCA, see Ekvall and Weidema, 2004), we have chosen to include the agricultural stage in milk and other dairy products, by using the scenario without milk quotas supplied by the LCA Food database.

Agricultural production leads to carbon fixation in biomass, and part of this carbon is also retained in animals biomass. Carbon fixation has been taken into account as a negative (-) emission of CO_2, calculated from the elemental composition of food. This elemental composition is obtained from nutritional composition (dry mass, carbohydrates, protein, etc.) and the human excretion model by Muñoz *et al*. (2008).

Wholesale and retail

Data from the LCA Food database has been used for these processes, including transport distances.

Transport to home

According to national statistics (Ministerio de Agricultura, Pesca y Alimentación, 2006a), 71% of the population goes to the market on foot, 20% by car, and 7.2% using public transportation. The car and bus trips have been attributed the environmental burdens of fuel consumption, as described in Milà i Canals *et al*. (2007).

Home storage

Instead of allocating refrigerator and freezer energy demand on a product basis, the total electricity consumption per person per year has been calculated, as this approach requires less data and is considered to be much less uncertain. A combined refrigerator and freezer per household, with a capacity of 255 and 94 L, respectively is considered to have a power consumption of 2.7 and 8.2 Wh L^{-1} day^{-1}, respectively, as is included in the LCA Food database. This appliance is assumed to be working all year round, and the average number of persons per household is 2.81 (Instituto Nacional de Estadística, 2007). This leads to an average electricity consumption of 190 kWh $person^{-1}$ $year^{-1}$.

Cooking

The environmental burdens of cooking are related to the energy used to prepare food. This has been quantified on a product basis, by defining a cooking scenario for each product category, and an energy use factor for each cooking mode. These energy use factors are obtained from Foster *et al*. (2006) - 3.5 MJ kg^{-1} for boiling; 7.5 MJ kg^{-1} for frying, 0.34 MJ kg^{-1} for microwaving; and 8.5 MJ kg^{-1} for roasting. Fifty percent of the energy used for cooking is assumed to originate from gas appliances, and the remaining 50% from electric appliances (with the exception of microwave ovens, which are only powered by electricity). The cooking scenario (Tab. 3) is based on the authors' judgement; it is therefore subject to an important degree of uncertainty.

Tab. 3: Cooking scenario.

100% raw or no cooking	Salted meat, lard, cured meat, other processed meat, all canned fish, milk shake, yoghurt, cheese, other dairy products, bread, biscuits and cakes, chocolate and cacao powder, sugar, margarine, cucumber, lettuce, tomato, leafy vegetables, processed potatoes, all fresh fruits, canned fruits and vegetables, frozen fruits and vegetables, sauces, olives, all nuts, all beverages.
100% microwaving	All milk.
100% frying	Knackwurst sausages, frozen potatoes.
100% boiling	Rice, pasta, coffee and infusions.
75% frying, 25% roasting	All fresh and frozen meat, all fresh and frozen fish, pulses.
50% boiling, 50% frying	Eggs, all seafood.
50% frying, 50% raw	All vegetable oils.
33% boiling, 33% frying, 33% roasting	Garlic, mushrooms, fresh potatoes.
25% boiling, 25% frying, 25% roasting, 25% raw	Onion, pepper, other fresh vegetables.
50% boiling, 25% frying, 25% roasting	Cabbage, green beans, asparagus.

Solid waste

The amount of kitchen waste produced has been calculated on the basis of the edible portion of each product category (Ministerio de Agricultura, Pesca y Alimentación, 2006b). The percentage of non-edible portion ranges from 0% for beverages and other products to 32% for fish and seafood. The weighed percentage for the functional unit is 9%, or 71 kg $person^{-1}$ $year^{-1}$. This amount of waste is assumed to be entirely generated before cooking; therefore this mass of food is not taken into account in the cooking stage.

According to urban waste management statistics (Ministerio de Medio Ambiente, 2008), around 70% of the organic waste generated in Spain is treated in composting plants, whereas the remaining fraction is mainly landfilled. Both Composting and sanitary landfilling have been modelled with Ecoinvent data (Nemecek and Kägi, 2007; Doka, 2003).

Human excretion and wastewater treatment

The main input to the human excretion model by Muñoz *et al.* (2008) is the nutritional composition of food, as shown in Tab. 4. This composition constitutes the average for the 99 individual food items included in the study, weighed according to the amount ingested. Individual food compositions have been obtained from nutritional data tables (Martín-Peña, 1997).

Tab. 4: Nutritional composition of the Spanish diet.

Constituents	**% in weight**
Water (g)	73.8
Protein (g)	4.6
Fat (g)	5.9
Carbohydrates (g)	12.9
Fibre (g)	0.92
Alcohol (g)	0.92
P (g)	0.071

With regards to wastewater treatment, due to the lack of data on the type of treatment plants, only 50% of the wastewater is considered as being treated with nutrients removal within the wastewater treatment plants (WWTP); the remaining 50% receives secondary treatment only within the WWTPs. Sixty five percent of excess sludge is sent to agricultural applications (Ministerio de Medio Ambiente, 2008), while the remaining 35% is mostly sent to landfills. Land application is included in the excretion model, while sludge landfilling has been included using the Doka model (2003).

Results

Fig. 1 shows the LCIA results per functional unit, i.e. feeding an average Spanish person for a year. The figure shows the absolute contribution of four life cycle stages, plus carbon fixation in Fig. 4a. In addition, the contribution of several food items and processes described in the previous sections can be seen.

The net GWP related to feeding an average Spanish citizen for a year (Fig. 1a) is 1.56 tons CO_2-eq. This figure is dominated by the food production stage, which also includes distribution and retail. Highlighted contributions are those of meat products and dairy (65% of the total GWP for food production). Nevertheless, human excretion and WWTP is the second most important life cycle stage (24% of total CO_2-eq. releases), due to carbon releases in respiration, wastewater treatment, sludge disposal, and, although not shown in the figure, to auxiliary materials like toilet paper, soap, and tap water. Home processes are also responsible for a relevant contribution to the life cycle, while waste management and home transport are almost negligible. Another interesting aspect to be observed in Fig. 1a is how the carbon balance is closed within human excretion, since carbon which is fixed in biomass is mostly released during that stage.

EP (Fig. 1b) is also dominated by the food production stage, where meat products alone are responsible for 60% of the total emissions in this part of the life cycle. Again, human excretion and WWTP is the second most important stage (22% of total EP). This importance is mainly related to the release of nitrogen and phosphorus compounds in the treated sewage. The contributions of transport to home, storage and cooking, and solid waste management are negligible.

The overall PEU per citizen per year is 16.4 GJ, with the most important contribution (Fig. 1c) taking place during food production, and with several food groups being important contributors. Home processes, i.e. storage and cooking, are very relevant from an energy perspective (30% of total PEU), using an amount of energy equivalent to half that used to produce and distribute food. The importance of human excretion and WWTP is lower in this indicator (8% of total PEU), although higher than that of transport to home and kitchen waste management.

Finally, in AP (Fig. 1d) food production is again the most important stage, with meat and dairy appearing as the most polluting food groups. Besides food production, only home storage and cooking have a relevant contribution. Human excretion and WWTP are negligible in this impact category (3% of total AP).

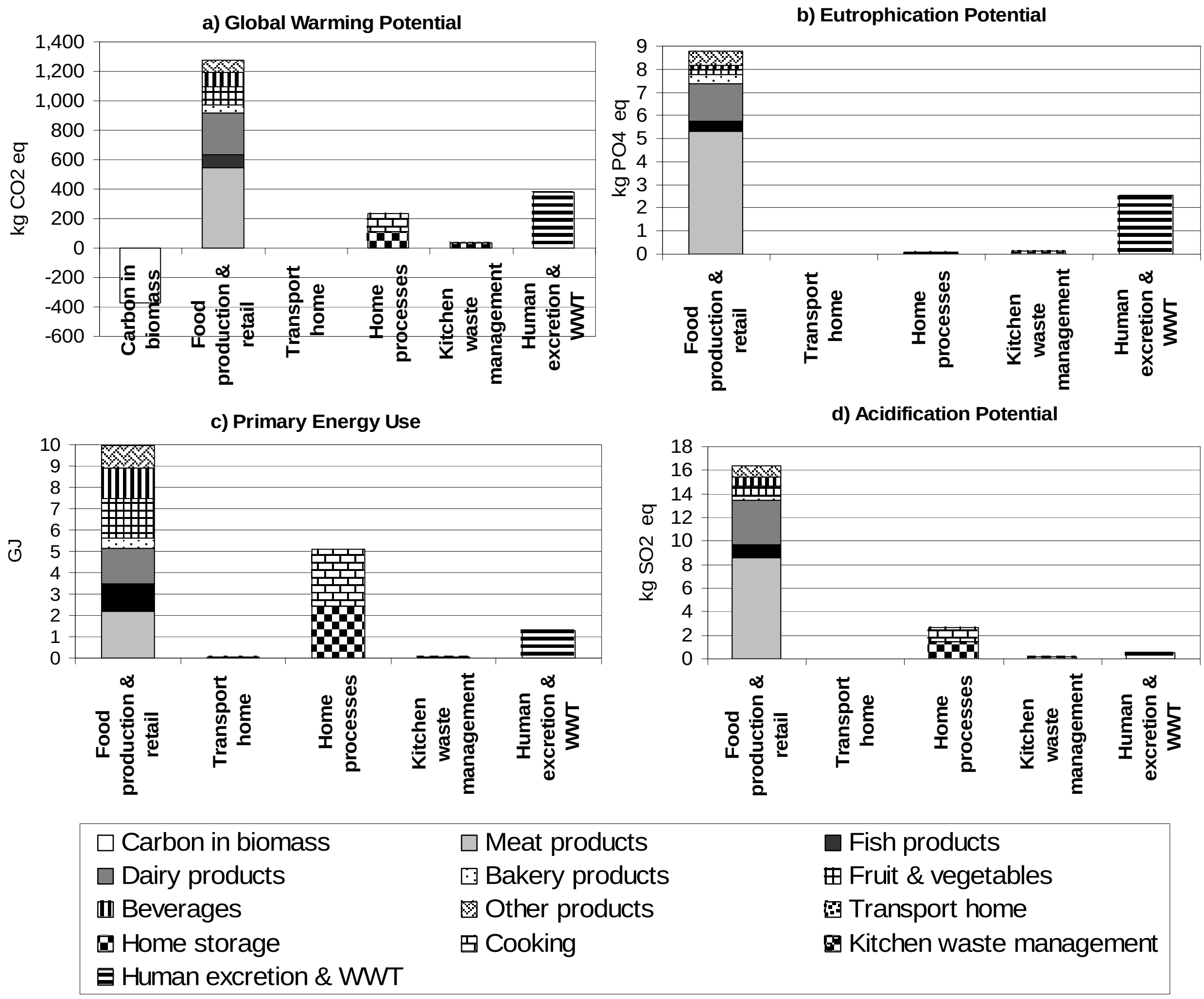

Fig. 1: Life Cycle Impact Assessment results per functional unit.

Discussion

The results obtained can be compared to those from previous studies. GWP was studied by Carlsson-Kanyama (1998) for different dietary models, leading to greenhouse gas emissions in the range of 420-3800 kg CO_2-eq. person^{-1} year^{-1}, while in our study the overall GWP is 1560 kg CO_2-eq. person^{-1} year^{-1}, including the excretion stage. Kramer *et al.* (1999) calculated GWP of food supply using an input-output approach, leading to emissions of 2800 kg CO_2-eq. household^{-1} year^{-1}, but the number of persons per household is not stated. With our results, a Spanish household would be responsible for 4370 kg CO_2-eq. year^{-1}, a figure substantially higher; the different approaches used (input-output/process LCA) could explain the differences in these two studies. Carlsson-Kanyama *et al.* (2003) found that PEU for Swedish food consumption may be in the 6.9-21 GJ person^{-1} year^{-1} range, while the figure obtained in our study is 16.4 GJ person^{-1} year^{-1}. Concerning contributions of food items, many studies highlight animal food as a critical issue (Kramer *et al*. 1999; Carlsson-Kanyama, 1998; Jungbluth *et al.* 2000), something in accordance with our study, especially in GWP, EP and AP.

It looks as though these previous studies achieved similar results without including human excretion within their system boundaries. In GWP, this is due to the fact that most carbon emissions in the excretion stage are offset by carbon fixation from photosynthesis, resulting in an almost neutral carbon balance. In the case of PEU, human excretion has a rather low contribution to the overall life cycle; therefore, omitting this stage does not change the overall picture by very much. However, this does not hold true for EP, an impact category not assessed in the cited studies, where human excretion is a life cycle stage of the utmost importance, as pointed out by Sonesson *et al.* (2004) and Muñoz *et al.* (2008). In Sonesson *et al.* (2004) the contribution of nutrients and chemical oxygen demand was assessed for different food items, resulting in total life cycle contributions to EP as high as 70% for apples and 55% for bread. In Muñoz *et al.* (2008), a contribution of 45% is found for broccoli. In our case study, the integrated contribution from the whole Spanish diet is a relevant 22%.

Nevertheless, our study has many limitations. First of all, the data gaps in the production stage, where rather detailed statistics are available, but limited product inventories in databases such as the Danish LCA Food. Inventory data for food production in the Spanish context is even scarcer. Furthermore, the LCA Food database has been built following a consequential approach to system boundaries, allocation and data selection, whereas our study can be labelled as attributional. Although we have dealt with this problem by adapting some datasets (dairy production without milk quotas, average mix for electricity production instead of the marginal production technology), the use of this database could lead to biased results. In addition, environmental burdens of packaging have been excluded. Although Jungbluth *et al*. (2000) found this aspect of minor importance for meat and vegetables, it could be important for other products such as beverages. Data for the cooking process, namely the share of cooking modes and the share of electric/gas cooking appliances, are very uncertain in this case study, but no alternative data were found; nevertheless, a sensitivity analysis dealing with the share of frying, boiling and roasting showed only little changes when the overall results were concerned. Finally, the amount of kitchen waste is probably underestimated, since in addition to the non-edible portion of food, leftovers and food gone bad have been neglected. According to WRAP (2008), UK households throw away as many as 1/3 bought food items.

Conclusion

Human excretion as a life cycle stage has been found to be important in the average Spanish diet, especially in relation to Eutrophication Potential. This is due to the emissions of nutrients in treated sewage. After food production, human excretion appears as the most important source of emissions in Eutrophication Potential and Global Warming Potential, while in Primary Energy Use and Acidification Potential it is not an important stage. In all these impact categories, food production is the most important life cycle stage, highlighting especially meat and dairy products.

Human excretion should not be overlooked in LCA studies dealing with dietary shifts, since the emissions related to this life cycle phase are different when different food items are considered. Neither should it be omitted in attributional studies aimed at identifying the life cycle hotspots of a given food product or diet.

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Assessment of aggregated indicators of sustainability using PCA: the case of apple trade in Spain

J. Soler-Rovira[1] and P. Soler-Rovira[2]
[1]Departamento de Produccion Vegetal: Fitotecnia, Escuela Universitaria de Ingenieria Tecnica Agricola, Universidad Politecnica de Madrid, Ciudad Universitaria s/n, 28040-Madrid, Spain, jose.soler@upm.es
[2]Instituto de Ciencias Agrarias, Centro de Ciencias Medioambientales, C.S.I.C., Serrano 115 dpdo., 28006-Madrid, Spain.

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Abstract

Environmental, economic and social impacts of intensive agricultural production, but also those regarding to international trade of fresh apples in Spain, were studied by the multivariate statistical method of principal components analysis (PCA). Environmental indicators were developed for 36 countries using life cycle analysis of apple cultivation and transport, weighting the results on a global or local scale. Economic and social indicators were also calculated considering macro and microeconomic aspects and also farm or society characteristics. PCA was applied to each set of indicators and aggregated indices were computed for each dimension of sustainability with the results of the analysis. The selected indicators explained with good agreement the differences in sustainability between countries and the synthetic indices ranked them all. Some of them showed a high relative sustainability, while other presented low values, due to low environmental, economic or social sustainability values of the aggregated indices.

Introduction

In recent years, international trade of fresh fruits and vegetables has also increased due to market liberalisation and technical development of agricultural practices, conservation processes and transport facilities. There are a variety of complex environmental, economic and social impacts regarding international trade. Those impacts arise at either global or local scales, pertaining to issues as energy consumption, emission of pollutants, degradation of natural resources, land-use changes, etc. By the other hand, economic growth takes place in exporting countries, but wealth often shows and unequal distribution between the populations (Wurtenberger *et al.*, 2006). Consequently, there is a need to assess the environmental, economic and social impacts of intensive agricultural production but also those regarding to international trade.

Spain is a big fruit producer with approximately 13 million tonnes in 1,300,000 ha of cultivated land, which represents almost 30% of the harvested area of fruits in the EU (FAO, 2008). The trade flows of agricultural products with other countries show a balanced state since the imports equals the exports. The main exported agricultural product is the fresh fruit (4,300 million euro in 2005), but Spain also imports fruits for 1,300 million euro (MAPA, 2006). Apple production and trade is a good case study because it accounts for 14% of total fruit consumption in Spain, and 29% of this quantity is imported from other countries (MAPA, 2006). Spanish imports of apple have been increased in the last years and the main origins are France, Italy, Germany and Portugal in the EU, and Chile and Argentina in South America (FAO, 2008). Imports from France have decreased in the period 2000-05, but the other origins show an increasing trend, except Portugal that shows a flat evolution.

The sustainability of this apple trade may be assessed by a number of indicators that reveal the impacts of their cultivation and transport from producer countries to the Spanish market. These indicators should be useful for policy-makers at the roundtables where trends are monitored and sustainable trade policies are introduced and evaluated. Even more, they can be helpful product information for consumers and their associations towards a consumption trend that accomplish a set of sustainability-sensitive criteria (Levitan, 2000). Indicators should be synthesized into an appropriate indicator that contains a lot of information but, at the same time, it is easy to understand by the end-users (policy-makers, consumers, etc.). Aggregated indicators help to communicate the information succinctly and make easier to distinguish patterns in the data by formalizing the aggregation process that is often done implicitly, subjectively and intuitively (Jollands *et al.*, 2004). Nevertheless, it is necessary to consider the potential limitations of the synthetic indicators since they may mask and simplify the complexity of environmental, social and economic systems.

In the development of aggregated indicators problems arise when the indices that build up the indicator and the weights of each index have to be selected. Therefore, principal components analysis (PCA) can be used as an objective approach to choose the indices that show higher variability within the studied observations and to set the weights as a function of the explained variance (Jollands *et al.*, 2004). However, PCA is limited to ex-post analysis and it is not an appropriate tool for prospective investigations. Additionally, this analysis allows making an internal sustainability evaluation between countries, giving a relative value of sustainability.

PCA have been used in several studies that include large sets of data, i.e. ecology and water quality, landscape characterization, pesticide screening or food quality. PCA has been applied to select proper and representative variables that could explain the variability included in the original data. The usefulness of PCA have been demonstrated to select environmental (Yu *et al.*, 1998), energy intensity (Bernard and Cote, 2005), eco-efficiency (Jollands *et al.*, 2004) and agri-environmental indicators (Soler-Rovira and Arroyo-Sanz, 2003; Soler-Rovira and Arroyo-Sanz, 2004). For example, the latter authors selected nutrient management indicators and classified the Spanish provinces and districts applying cluster analysis to the results from the PCA. Finally, synthetic indicators have been aggregated by PCA for data obtained in studies on sustainable agricultural systems (Sands and Podmore, 2000), irrigation schemes (Rodriguez-Diaz *et al.*, 2008), poverty and human development (Antony and Visweswara Rao, 2007) and sustainable development and environmental quality (Castro, 2002; Jha and Murthy, 2003; Escobar, 2006). However, the methodology used to build up the aggregated indicator differs between authors and none of them have applied the aggregation with PCA combined with life cycle analysis on sustainability of production and trade.

The aim of this work is to assess the sustainability of apple production and trade flows in Spain by the development of aggregated indexes obtained by multivariate analysis (PCA) of individual indicators (economic, social and environmental).

Method

The methodology used is PCA, so first of all this multivariate statistical tool is briefly described in this section. Secondly, the characteristics of apple trade in Spain are evaluated and the main apple producing countries are selected. The next steps are to choose a set of indicators that can be used to characterize the sustainability of the environmental, economic and social dimensions of apple trade. Lastly, all the selected indicators are synthesized in a aggregated index of sustainability. All these steps are extensively described below.

A brief description of principal components analysis

PCA is a statistical multivariate methodology used to study large sets of data. This method reproduces a great proportion of variance among a big number of variables by using a small number of new variables called principal components (PCs). The PCs are linear combinations of the original variables, and the analysis of multidimensional data is simplified when these are correlated (Judez, 1989). The first PC explains maximum variance between data, while the second component is a new combination of the original variables being orthogonal to the first component and explaining the second largest value of variation among observations, and so forth. The absorption of variance in each component is computed with the so-called eigenvalues. One property of the PCs is that they are uncorrelated between them, and then each component is measuring a different dimension in the data.

High absolute values of loadings of the variables (i.e. indicators) on the PCs imply that the indicator has a large bearing on the creation of that component. Thus, the most important indicators in each component, that best explain variance; will also be more useful in explaining variability between observations (i.e. countries). Each component will be a linear combination of indicators (variables) multiplied by their loadings on that component. Observations (countries) will have coordinates in each axis or component, computed with the standardized value of each variable (zero mean and unit variance) for that observation using the linear combination of variables with PCs obtained in the analysis.

Apple trade in Spain

Twenty most important apple exporters of the world and other 16 countries that have exported apples to Spain in the last 10 years have been selected as the observations set. Data of apple exports and imports are from FAO (2008). The 36 selected countries are shown in Tab. 3.

Environmental indicators

The environmental dimension was analysed considering the crop production and the transport of the apples. Agricultural practices were assessed searching information about fertilization, irrigation and yield of apple orchards, i.e. FAOSTAT (FAO, 2008), International Fertilizer Industry Association (IFA, 2008) and Water Footprint of Nations (Waterfootprint, 2008). The agricultural impact was calculated for 1 hectare (ha) of orchard and for 1 kilogram (kg) of fresh apples (just dividing by yield). Irrigation impact was considered as the water requirements of the crop for one year (Waterfootprint, 2008) in m^3 per ha or per kg of apple. Fertilization impact was assessed by computing emissions and inputs during manufacturing of fertilizers, so data per kg N, kg P_2O_5 and kg K_2O manufactured were used. A nutrient balance was carried out in apple orchards, considering atmospheric N emissions from fertilizers (NH_3, N_2O and NO) with EMEP methodology (EEA, 2004) and nitrate leaching as the mean of a constant value of 16% of N inputs in fertilizers (Neilsen and Neilsen, 2002) and the result of the balance $N_{leaching}= N_{fertilizers}-N_{crop\ uptake} - N_{gaseous\ emissions}$, when it was positive. The inventory of transport was done with the distance from production zones to Madrid (Spain) and using lorry and ship emissions per t km transported. Road transport by lorry was considered from countries in continental Europe and distance was computed by data from ViaMichelin (2008). Sea transport by ship was considered from the other countries, computing distance from the main port of the country to Algeciras or Valencia in Spain (the two main ports for fruit trade), via Panama, Suez or Gibraltar (Sea Distances, 2008). Lorry transport from the ports to Madrid was also taken into account. Life cycle analysis of transport data was done for 1 kg of fresh apples from each country.

Ten impact categories were considered: global warming, acidification, eutrophication, human toxicity, ecotoxicity in fresh water, photochemical oxidants formation, energy use, water resources use, abiotic resources depletion and land use. Characterization factors for each category were use from CML-IA (2004). World in 1995 normalization factors were used (Van den Berg *et al.*, 1995; Huijbregts *et al.*, 2003; CML-IA, 2004).

Normalized values of the LC analysis of each impact category were added up for crop production (LCAcrop indicator) and for transport (LCAtransport indicator), and the sum of those two was an overall potential environmental impact indicator (LCAtotal). Other two indicators were calculated considering a local and a global geographical scale. The impact of apple production over local population and ecosystems was calculated per ha of cultivated land, considering that the main impact categories were toxicity for human population, depletion and pollution of water resources and land use and occupation for agriculture. A multi-criteria analysis was carried out using analytical hierarchy process (AHP) (Saaty, 1990). The ten studied impact categories were ranked in a sequence from more to less relative importance at local scale: Human toxicity = water resources use = eutrophication = ecotoxicity fresh water = land use > acidification = photochemical oxidants formation > energy use = abiotic resources depletion > global warming. Based on these assumptions, the respective weights for each of the ten impact categories were calculated according to the AHP procedure. These weights were applied to the crop LC analysis and a local impact indicator was considered (LCAlocal).

The global impact of apple production and trade was calculated per kg of apple, considering that the main impact categories were climate change, energy use and depletion of natural resources. An AHP analysis was carried out sorting the ten impact categories in a sequence from more to less relative importance at global scale: Global warming > energy use = abiotic resources depletion = land use > water resources use = eutrophication = ecotoxicity fresh water > acidification > human toxicity > photochemical oxidants formation. The weights were calculated as for the local scale and they were applied to the crop and transport LC analysis and a global impact indicator was considered (LCAglobal).

Other environmental indicators were also calculated pertaining to particular aspects of environmental impacts, as productive land requirements, use of resources or emissions during the apple life-cycle. Ecological footprint was determined considering the yield of the orchards and the CO_2 emitted during fertilizers manufacturing and apple transport. Arable land and sink forest land for CO_2 were calculated and equivalence factors (Wackernagel *et al.*, 1999) were applied to determine the ecological footprint, i.e. m^2 of land required per kg of apples. Carbon footprint was computed as the kg of CO_2 equivalent per kg of apples emitted during cultivation and transport. Water footprint was calculated considering the yield and the water requirements in each country (m^3 ha^{-1}) in L of water per kg of apples. Energy footprint was determined as the energy used in fertilizer manufacturing and apple transport (MJ kg^{-1}). Reactive nitrogen released to the biosphere during fertilization and NO_x emitted in fertilizer industries and apple transport were also calculated (g N kg^{-1}).

A synthetic environmental indicator was calculated by PCA using a matrix of 18 variables x 36 countries. The initial set of environmental indicators included a large set of variables in order to firstly investigate which of them showed higher variability within the studied observations and correlation within them, that is strength of PCA, although some of them should explain redundant information. Thus, fertilization rates (kgN/ha, kgP/ha and kgK/ha), fertilizers per unit of apple produced (kgN/kg, kgP/kg and kgK/kg), water requirements (m^3/ha), water footprint (L/kg), transport distance (km), ecological footprint, energy footprint, carbon footprint, reactive nitrogen, and LCA values (LCAcrop, LCAtransport, LCAtotal, LCAglobal and LCAlocal) were included. Before developing the PCA, all the variables were signed as positive or negative in order to make them unidirectional (Jha and Murthy, 2003). PCA was performed with STATGRAPHICS software, standardizing data to zero mean and unit variance. Eigenvalues and the amount of variance explained by each principal component (PC) were calculated. The number of components retained in the analysis was assessed by Cattel's scree plot, which indicates that we should retain i components because, after the i+1 component, the plot becomes flat, corresponding to eigenvalues lower than one. The value of the eigenvectors and loadings of variables with PCs were computed. Coordinates of each country with each axis were determined. The aggregation of data into a single environmental sustainability index was calculated as:

$$PCA_{environmental}(i) = \frac{\sum_{k=1}^{j} F_{ki}\sqrt{\lambda_k}}{\sum_{k=1}^{j} \sqrt{\lambda_k}} \qquad i = 1,...,36(countries) \qquad [1]$$

Where, F_{ki} is the coordinate of the country i in the component k (and j components are retained) and λ_k is the eigenvalue of the component k. This index should give information about the relative value of environmental sustainability between the studied countries, taking into account that LCA gives an estimation of potential impacts.

Economic indicators

Economic dimension was analysed considering micro and macro economic aspects of apple trade. Micro economic level was studied at farm scale, so productivity, yield stability and yield sustainability were calculated as fundamental properties of farming systems (Marten, 1988). These indicators were computed as Tab. 1 shows. Macro economic level was assessed by nine indicators. One studied aspect was the positive effects of exportation of agricultural products as the returns obtained by apple exports, with a high market share and competitiveness. On the other hand, some negative aspects would arise as market oligopoly with an export-oriented farming with apple cultivation as a monoculture. Other negative factors could be the decay of exportation prices (Barriga, 2003) or the decreasing and abandonment of apple cultivated area. Other interesting indicator is called globalization in the sense that in some countries a great volume of imports and, at the same time, exports of apples exists, so the national market is decidedly open to the global market. All the statistical data were obtained from FAO (2008).

A synthetic economic index was calculated by PCA using a matrix of 12 variables x 36 countries (Tab. 1). All the variables were signed as positive or negative in order to make them unidirectional (Jha and Murthy, 2003). PCA was performed as described above and the aggregation of data into a single economic sustainability index ($PCA_{economic}$) was computed as in equation [1].

Tab. 1: Initial set of economic and social indicators used in the analysis.

Economic dimension	
Indicator	Calculation
Productivity	Average yield of apple in the period 1996-2003.
Yield stability	Coefficient of variation of apple yield during 1996-2003.
Yield sustainability	s_i/Y_i; where: s_i: slope of yield over the period 1996-2003 in country i. Y_i: average yield in country i during that period.
Exports value	$(AEV_i/AgEV_i)$*100; where: AEV_i: average apple exports value in country i in 2000-05 period. $AgEV_i$: average agricultural exports value in country i in 2000-05 period.
Market share	(AEC_i/AEW)*100; where: AEC_i: apple exports in country i over the period 1996-2004. AEW: total apple exports during that period in the world.
Competitiveness	Es_i/E_i; where: Es_i: slope of apple exports in country i over the period 1996-2004. E_i: average exports in country i during that period.
Oligopoly	MS_i-AMS; where: MS_i: market share of country i. AMS: average market share of each country if all world exports where fairly distributed.
Export-oriented farming	(AEA_i/TAA_i)*100; where: AEA_i: area of exported apples in country i in the period 1996-2004. TAA_i: total area of apple in country i in the period 1996-2004.
Monoculture	(AA_i/TFA_i)*100; where: AA_i: area of cultivated apples in country i in the period 1996-2004. TFA_i: total fruit cultivated area in country i in the period 1996-2004.
Exports price	Slope of apple exports prices over the period 1996-2004 in each country.
Abandonment apple area	Slope of apple cultivated area over the period 1986-2006 in each country.
Globalization	$[(AI_i+AE_i)/AP_i)]$*100; where: AI_i: apple imports in country i in the period 1996-2004. AE_i: apple exports in country i in the period 1996-2004. AP_i: apple production in country i in the period 1996-2004.
Social dimension	
Indicator	Calculation
Income stability	Coefficient of variation of orchard income (yield x price) during 1996-2003.
Income trend	Slope of income over the period 1996-2003.
International justice	APP_i-APP_{av}; where: APP_i : apple producer prices in country i during 1996-2003. APP_{av}: apple producer prices of the major 20 exporting countries during that period.
Market-farmer equity	(APP_i/ACP_{Sp})*100; where: APP_i: apple producer price in country i during 1996-2003. ACP_{Sp}: apple consumer price in Spain in 2007/08.
Fruit deficit	FVC_i-$FVCR_{WHO}$; where: FVC_i: fruits and vegetables consumption per capita in country i in the year 2003. FVC_{WHO}: fruits and vegetables consumption recommended by WHO (400 g day^{-1}).
Fruit diversity	Shannon index of fruit consumption per capita in year 2003.
Food waste	AW_i/AS_i; where: AW_i: apple waste in country i in year 2003. AS_i: apple supply in country i in year 2003.
Own supply	AS_i/FVC_i; where: AS_i: apple supply per capita in country i in year 2003. FVC_i: fruits and vegetables consumption per capita in country i in year 2003.

Social indicators

Social dimension was analysed considering farm level and society level. Farm level was assessed with four indicators and, first of all, farm income stability and income trend were calculated (Tab. 1). Market-farmer equity calculates the percentage of the final price (paid by consumer for 1 kg of apples) that farmers receive. International justice was also computed as the difference in revenues between apple in one country and average revenues from apples of the 20 major apple exporting countries (Wurtenberger *et al.*, 2006). Social aspects are related to fruits and vegetables deficit in the diet from the minimum recommended by World Health Organization (i.e. 400 g $capita^{-1}$ day^{-1}) and diversity of types of consumed fruits. Other social aspects are food waste and own supply of apples. All the statistical data were obtained from FAO (2008), except apple price paid by consumers in Spain (MAPA, 2008).

A synthetic social index was calculated by PCA using a matrix of 8 variables x 36 countries (Tab. 1). All the variables were signed as positive or negative in order to make them unidirectional (Jha and Murthy, 2003). PCA was performed as described above and the aggregation of data into a single social sustainability index (PCA_{social}) was computed as in equation [1].

Sustainability index

A synthetic sustainability index was built up considering environmental, economic and social indicators studied in the previous sections. Eleven environmental indicators were selected from the initial set of 18, considering those that showed a high correlation coefficient with the $PCA_{environmental}$ aggregated index, discarding also those that gave redundant information (e.g. fertilization rates or LCA_{total} indicator). The same procedure was carried out to select 9 economic and 8 social indicators. PCA was performed using a matrix of 28 variables x 36 countries. Aggregation was done as showed in equation [1] and a $PCA_{sustainability}$ index was computed.

Results and discussion

Principal components analysis for the environmental indicators is shown in Tab. 2. Five principal components were retained and they explained 93.5% of the total variance of the data. The first component (PC1) is highly correlated with five indicators that describe ecological footprint (EF), water footprint and LCA results for crop, total system and global scale. These indicators are related to yield and water consumption of apple orchards. The second PC is correlated with LCA in transport and related indicators as carbon and energy footprints and distance covered. PC3 is correlated with nitrogen and potassium fertilization, PC4 with impacts at a local level (per ha) and PC5 with phosphorus fertilization.

These indicators will explain with good agreement the differences in environmental sustainability between countries, and the coordinates of each country with each component will built up the synthetic environmental index, weighted with the eigenvalues of each component. The resulting index for each country and the corresponding ranking between all, are shown in Tab. 3. Twenty countries show positive values, thus higher than the mean (that is zero). The other 16 countries show negative values. France, Netherlands, Belgium and Switzerland are in the best positions in the relative hierarchy of environmental sustainability, while China, Cyprus, Iran and Korea are in the lower part of the ranking.

Another five principal components were retained in the analysis of the economic dimension (Tab. 2). The first PC would be defined as international trade, as it is positively correlated with market share and the subsequent revenues from exports, and negatively with market oligopoly. The economic sustainability index is positive in 17 countries and the other 19 show values under the mean. The first positions in the relative ranking are held by Chile, USA, Italy and France, while Finland, Latvia, Cyprus and Morocco are located in the last positions.

Social indicators were explained by four PCs. The first PC shows high loadings with market-farmer equity and international justice related to apple prices, and with food waste. The second PC is highly correlated with farm income and fruit deficit in population's diet. Regarding the aggregated social index ranks Cyprus, UK and Switzerland in the first positions of the 25 countries with positive value, while 11 show negative values as Moldova, Chile and Latvia that show the lowest values.

Tab. 2: Principal component analysis for environmental, economic, social and sustainability indices.

Environmental indicators

PCs retained	PC1	PC2	PC3	PC4	PC5
Eigenvalues	7.34	4.41	3.00	1.84	1.18
Variance absorption (%)	38.6	61.8	77.6	87.3	93.5
Correlated indicators (loadings)	EF (0.95) LCA_{crop} (0.94) WaterF (0.94) LCA_{total} (0.93) LCA_{global} (0.91)	$LCA_{transport}$ (0.99) EnergyF (0.96) CarbonF (0.95) Distance (0.87)	kgK/ha (0.92) kgK/kg (0.91) kgN/kg (0.73) kgN/ha (0.67) $N_{reactive}$ (0.63)	LCA_{local} (0.96) m^3/ha (0.96)	kgP/ha (0.88) kgP/kg (0.85)

Economic indicators

PCs retained	PC1	PC2	PC3	PC4	PC5
Eigenvalues	3.44	1.90	1.60	1.30	1.10
Var. absorption (%)	28.6	44.5	57.8	68.6	77.4
Correlated indicators (loadings)	Oligopoly (-0.95) Market share (0.95) Exports value (0.60)	Yield sustainability (0.82) Globalization (0.82) Yield stability (0.60)	Exports price (0.80) Abandonment (-0.69) Competitiveness (0.66)	Ex. oriented (-0.83) Productivity (0.82)	Monoculture (0.91)

Social indicators

PCs retained	PC1	PC2	PC3	PC4
Eigenvalues	2.18	1.70	1.18	1.00
Var. absorption (%)	27.3	48.5	63.2	75.8
Correlated indicators (loadings)	Equity (-0.95) Int. justice (0.72) Food waste (0.50)	Income trend (0.82) Fruit deficit (0.60) Income stability (0.56)	Own supply (0.92)	Fruit diversity (0.88)

Sustainability index

PCs retained	PC1	PC2	PC3	PC4	PC5	PC6	PC7
Eigenvalues	6.63	5.45	3.60	2.90	1.80	1.48	1.16
Var. ab. (%)	23.7	43.1	55.9	66.3	72.8	78.0	82.2
Correlated indicators (loadings)	LCA_{crop} (0.94) EF (0.93) WaterF (0.93) LCA_{global} (0.89) Productivity (0.80) Int. justice (0.64) Income st. (0.62) Yield st. (0.58) Exp. price (0.58)	$LCA_{transport}$ (0.96) CarbonF (0.96) EnergyF (0.94) Distance (0.85) Nreactive (0.62)	LCA_{local} (0.92) m^3/ha (0.92) Fruit diversity (-0.46)	Income trend (0.85) Yield sust. (0.84) Globalization (0.77) Equity (-0.57)	Market share (0.91) Oligopoly (-0.91)	Exports value (-0.83) Food waste (0.81) Export oriented farming (0.50)	Fruit deficit (0.74)

The results of the PCA for the 28 selected indicators (11+9+8) in order to develop a sustainability index are shown in Tab. 2. Seven principal components were retained and they explained 82.2% of the total variance of the original data. The first component (PC1) shows high loadings with a combination of environmental, economic and social indicators. The environmental ones were ecological and water footprints and LCA results for crop and global scale, so the main environmental issues were related to water and land use per kg of apples produced. The economic dimension was explained by two microeconomic indicators (yield productivity and stability) and one macroeconomic (trend of export prices).. Social sustainability was related to apple price indicators (international justice and farm income). The second PC is only correlated with environmental indicators related to apple transport, as LCA added values of transport, the distance covered, and the related energy consumed and equivalent carbon emitted. Moreover, reactive nitrogen emissions are captured by this second PC, although they consider the agricultural and transport phases. The third component is positively correlated with impact al local scale, mainly produced by water use per ha, and negatively with diversity in fruit diet. The last four PCs capture economic and social issues. These indicators will explain with good agreement the differences in overall sustainability between countries. The coordinates of each country with each component will built up the synthetic sustainability index for that country, weighted with the eigenvalues of each component (Tab. 3). Twenty countries show positive values of the synthetic index, and the other 16 are below the mean and show negative values in the relative ranking of sustainability performed by the analysis.

Tab. 3: Values of the aggregated indices for sustainability and environmental, economic and social dimensions of the 36 countries studied. Relative ranking of each country is also shown.

Country	$PCA_{environ}$	Ranking	PCA_{econ}	Ranking	PCA_{social}	Ranking	$PCA_{sustainability}$	Ranking
Argentina	-1.494	27	0.427	14	-0.980	31	-0.671	22
Austria	3.007	6	0.757	11	0.651	11	1.976	7
Belgium	3.523	3	1.237	9	0.400	15	2.416	4
Brazil	-0.008	21	0.367	15	-0.914	30	-0.846	24
Canada	1.819	13	-0.138	19	0.294	18	1.738	10
Chile	-0.620	25	2.789	1	-2.124	35	-1.151	26
China	-10.708	36	1.512	5	-0.733	29	-3.599	35
Cyprus	-9.614	35	-1.839	34	1.240	1	-3.476	33
Czech R.	2.875	7	-0.366	22	0.112	22	0.849	17
Denmark	1.682	15	-1.060	31	0.773	7	1.147	12
Finland	-1.799	28	-4.149	36	0.285	19	-2.314	32
France	3.875	1	1.987	4	0.147	21	2.769	1
Germany	2.061	12	0.708	12	0.373	16	0.957	16
Greece	1.428	18	-0.722	27	0.853	6	0.754	18
Hungary	-0.451	23	-0.403	23	-1.088	32	-0.684	23
Iran	-4.587	34	0.315	16	0.544	14	-2.147	30
Ireland	3.131	5	-0.524	24	0.162	20	2.183	5
Italy	2.070	11	2.284	3	0.547	13	1.934	8
R. Korea	-3.798	33	-0.189	20	0.667	10	-2.187	31
Latvia	-1.170	26	-3.804	35	-1.873	34	-3.510	34
Moldova	-2.937	32	-1.425	32	-3.063	36	-4.252	36
Morocco	-2.906	31	-1.604	33	0.065	23	-1.751	27
Netherlands	3.619	2	0.838	10	0.696	9	2.513	3
N.Zealand	-0.546	24	1.288	7	0.013	25	-0.924	25
Poland	-0.101	22	1.389	6	-1.744	33	0.221	20
Portugal	0.074	20	-0.548	25	0.623	12	0.646	19
Slovakia	2.138	10	-0.826	30	-0.207	27	-0.054	21
Slovenia	2.861	8	0.147	17	0.029	24	1.423	11
S. Africa	-2.569	30	1.240	8	-0.542	28	-1.985	29
Spain	1.690	14	-0.131	18	0.312	17	1.075	14
Sweden	0.112	19	-0.822	29	0.862	5	1.064	15
Switzerland	3.508	4	0.628	13	0.966	3	2.695	2
Turkey	1.596	16	-0.286	21	0.744	8	1.093	13
UK	2.671	9	-0.817	28	0.970	2	1.810	9
USA	1.437	17	2.357	2	0.950	4	2.096	6
Uruguay	-1.869	29	-0.614	26	-0.008	26	-1.808	28

Within the first 19 countries we can slightly separate two different groups. The first group include nine countries that show a positive value of the sustainability index and also show this positive value in the three previous computed indices pertaining environmental, economic and social sustainability. These countries show, in general, lower environmental impact in a global scale, a high productivity and a good justice for apple prices within the global market, although they have a tendency to monopolize it. This group includes USA and eight European countries: France, Switzerland, The Netherlands, Belgium, Austria, Italy, Slovenia and Germany. The second group is characterized by positive values of the aggregated sustainability index; they have a socio-environmental sustainability in the relative hierarchy ranked in the analysis, because only the environmental and social indices are positive and the economic index is negative. The environmental dimension is characterized by relative low environmental impact, particularly energy and carbon footprints, and high social sustainability, especially with regard to prices, income and waste indicators. The worse side is the low economic sustainability, represented by productivity and market share indicators. This group includes Canada, Turkey and eight European countries (Ireland, UK, Denmark, Spain, Sweden, Czech Republic, Greece and Portugal).

Another seventeen countries are in the low zones of the relative sustainability ranking established by the PC analysis. New Zealand and Iran show a positive socioeconomic sustainability index, with an important weight of apples in agricultural exports and relative good values for income and fruit consumption indicators. However, the environmental sustainability index is negative due to impacts at global scale and during the transport stage.

Argentina, Brazil, Chile, China and South Africa show a negative value of the sustainability index and reach only a positive value of the economic index. Environmental impact is relatively high for agricultural production and, above all, the transport phase due to the large distances to Spain. This impact is important at both global and local scale. Social aspects are eroded by low producer prices and deficit of fruit consumption. The economic advantages that show these countries are the market share for apples, but they tend to control it. Poland would be included with these economic sustainable countries, although it shows a positive sustainability index.

Slovakia is characterized by a positive environmental sustainability index due to global and local relative low impacts. However, economic aspects are shorted by low yield sustainability; and social dimension is deficient with regard to income trend of farmers and fruit consumption of the population. Positive social sustainability index is achieved by Cyprus, Finland, Republic of Korea and Morocco due to a relative high equity and justice in apple prices. Economic sustainability indicators as productivity and apple monoculture should be improved. The environmental impact at a global scale is another bad indicator.

The less sustainable indices are shown by Hungary, Latvia, Moldova and Uruguay. The three dimensions of sustainability show negative values and they are in the lower part of the sustainability index ranking. Productivity and monoculture should be improved in the economic dimension. Equity and justice of producer prices should be enhanced and deficit of fruits and vegetables consumption should be reduced. Environmental impact at a global scale should decrease, and ecological and water footprints should be improved.

Conclusion

Principal components analysis is a good statistical tool to develop aggregated indicators in order to asses the sustainability of apple production and trade flows in Spain. This multivariate analysis can be used as an objective approach to select the most important indicators regarding economic, social and environmental aspects of apple production and trade. The aggregation of data yields a single index easy to understand and that contains a lot of information, and allows to make a ranking between studied countries. Then, the sustainability of apple trade may be assessed by synthetic indices and strengths and weaknesses of each country may be discerned, and improvements may be suggested by studying individual indicators. The results for the main producing countries of apples imported in Spain show that France and Italy have a high sustainability index, Spain and Portugal just have positive values for the social and environmental aspects, while Argentina and Chile showed only positive values for economic sustainability.

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Veggie versus meat – environmental analysis of meals in Spain and Sweden

J. Davis[1], U. Sonesson[1], D. Baumgartner[2] and T. Nemecek[2]:
1 SIK – The Swedish Institute for Food and Biotechnology, Gothenburg, Sweden,
jennifer.davis@sik.se
2 ART – Agroscope Reckenholz-Taenikon Research Station, Zurich, Switzerland

Keywords: LCA, meal, peas, pork, vegetarian

Abstract

The cultivation, processing, packaging and transport of food contribute significantly to the overall environmental impact caused by man. However, different food products have quite different impacts; it is commonly stated that production of vegetarian products is associated with far less environmental impact compared to products of animal origin. In this study, the magnitude of the environmental benefit of pea protein compared to animal protein was investigated. Since peas do not only provide protein but also other nutrients like carbohydrates, the functional unit of the study is a complete meal providing ca 35 g protein and 3 100 kJ (the proportion of energy coming from fat, carbohydrates and protein respectively are similar in the four compared meals). Indeed, the study shows that the vegetarian meal causes significantly less environmental impact than the meals with animal protein; eutrophying and green house gas emissions are between 40% to 80% lower. Concerning energy use the picture is however different, the vegetarian meal uses about the same amount of energy as the meals with animal protein, which is due to energy demanding processing in industry of the pea burgers.

Introduction

Protein is one of the essential parts in our nutritional intake. According to the Swedish food administration protein should stand for 10-15% of the overall energy intake. In the Western world, the consumption of protein is generally higher than the nutritional need. At the same time, the production of protein is very resource intensive, so a diet with surplus protein is likely to be more resource demanding than a more balanced diet. There are various studies of environmental impact associated with choice of diet. Carlsson- Kanyama (1999) focused on global warming, analysing different meals with similar content of protein and energy but where the ingredients differed, being either animal or vegetable, and “exotic” or locally produced. However, no process data were analysed. Baroni *et al.* (2006) compared three diets, conventional, vegetarian and vegan, with a sub-comparison that the raw materials were produced organically or conventionally. They concluded that a reduced meat consumption was beneficial for most environmental impact categories, mostly prominent for land use. The studied diets were based on real consumption and the LCA data used were aggregated and process data were scarce. There are also other studies focused on energy use or GWP associated with diets (Dutilh & Kramer, 2000; Kramer *et al.*, 1999) or giving a framework how to assess the environmental impact of diets, all using data based on aggregate levels. The aim of this paper is to compare the environmental impact of meals with pea protein and animal protein (pork) on a process based level, exploring the whole process chain from farm to fork, i.e. from the farm and up until the meal is ready for consumption at the household.

Method

The method is briefly described below, for a full report see Davis & Sonesson (2008). The software SimaPro (Pré, 2006) was used to perform the calculations.

Meals

Four meals with different sources of protein have been analysed. Two countries (Sweden and Spain) are explored to highlight how the results depend on surrounding systems, i.e. eight meals have been analysed:

1. ***SOY pork chop:*** Pork chop produced with conventional feed (SOY = pig feed based on soyabean meal imported to Europe and cereals), potatoes, raw tomatoes, wheat bread and water
2. ***PEA pork chop:*** Pork chop produced with alternative feed (PEA = pig feed based on peas, rape seed and cereals mostly grown in Europe and some imported soyabean meal), potatoes, raw tomatoes, wheat bread and water
3. ***Sausage partial PEA:*** Meal with partial replacement of pig meat by peas; a sausage in which 10 % of the animal protein is replaced by pea protein (pork produced with PEA feed), raw tomatoes, wheat bread and water
4. ***PEA burger:*** Meal with full replacement of meat by a pea burger (peas grown in Europe), accompanied by raw tomatoes, wheat bread and water.

In the Spanish scenario the peas, pork, wheat and potatoes are produced in Spain, whereas in the Swedish scenario the origin of these products is Germany, except for the potatoes which are cultivated in Sweden. The tomatoes come from Spain in both scenarios. The potatoes are either roasted in the oven (Spain) or boiled (Sweden). The pork chop, sausage and pea burger are fried in a frying pan in both cases.

Functional unit

Food benefits us in several ways; besides providing us with energy, proteins and vitamins, there are other noteworthy aspects, such as, pleasure, and even cultural and social identity. We have chosen the function of basic nutrient supply. ***The functional unit for this study is one meal served at the table in the household***. The meals all deliver an equal amount of protein and energy: ca 35 g protein and 3 100 kJ. See Tab. 1 for the quantity of each component in the meals. The proportions between proteins, fat and carbohydrates are within the recommendations on nutrient intake from the Swedish Food Administration.

Tab. 1: Amount of ingredients of each meal [g or ml per meal]

Meal	Pork chop/ sausage/ burger [g]	Potatoes, peeled [g]	Tomatoes [g]	Bread [g]	Water (mineral water in Spanish scen.) [ml]
SOY pork chop	100[1]	350	90	100	300
PEA pork chop	100[1]	350	90	100	300
Sausage partial PEA	225	-	90	140	300
PEA burger	275	-	90	80	300

1) Weight without bone but with 5 mm fat rind

System boundaries

The analysis starts with raw material production in agriculture including production of inputs such as fertilisers and fuels. All inputs of packaging materials for the products are included as is the waste management of the used packaging (Fig. 1). Production of electricity and heat as well as water used in the system is included. Electricity for storing and cooking in households is included, as well as all transports involved throughout the chain. Finally, the environmental impact from sewage treatment is included in the analysis.

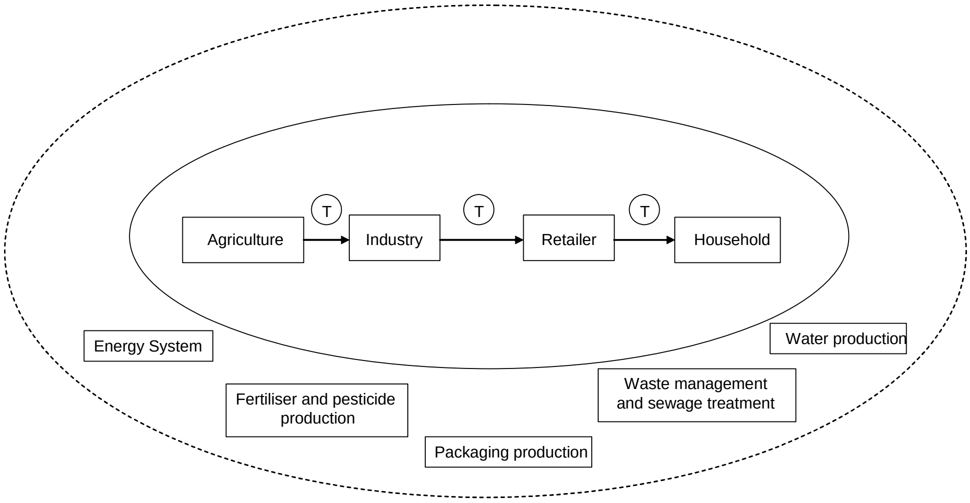

Fig. 1: Activities/processes included in the analysis

Data collection

Data on production of pork, wheat and peas have been collected from Baumgartner *et al.* (2008), data on industrial operations have been gathered from industrial contacts, and data on other materials and transport have been taken from literature and Ecoinvent (2004).

Type of LCA and allocation method

Attributional LCA has been used in the analysis (average data for background processes). The economic value of the outputs has been used to allocate the environmental burden between co-products (e.g. grinding of wheat which gives both flour and bran).

Results

The considered impact categories in this LCA study are use of renewable (biomass, wind and water) and non-renewable (fossil and nuclear) energy resources, global warming potential (time horizon of 100 years), photo oxidant formation potential (as precursors of ozone), stratospheric ozone depletion potential, eutrophication and acidification potentials. Energy use, global warming and eutrophication potential are discussed here; the full results are given in Davis & Sonesson (2008).

Fig. 2 and 3 show the use of primary energy (non-renewable and renewable) for the Spanish and Swedish meals respectively. The energy use for all four meals in each scenario is in the same order of magnitude, but the overall energy use is higher in the Spanish case, which is mostly due to the energy required in the household to oven bake the potatoes (in the Swedish case the potatoes are boiled). Moreover, in the Spanish scenarios, 1.3 MJ is required to produce the plastic bottle for the mineral water (included in ‘other’), and the contribution from the pig farm is also higher compared to the Swedish meals.

The reason why the pea burger meal is as high as the other meals is that we have assumed the pea burgers are sold as a frozen product, hence a lot of energy is used for freezing it in industry and then storing it in a freezer both at the retailer and at the consumer (there is also more energy needed for frying the burgers at the household).

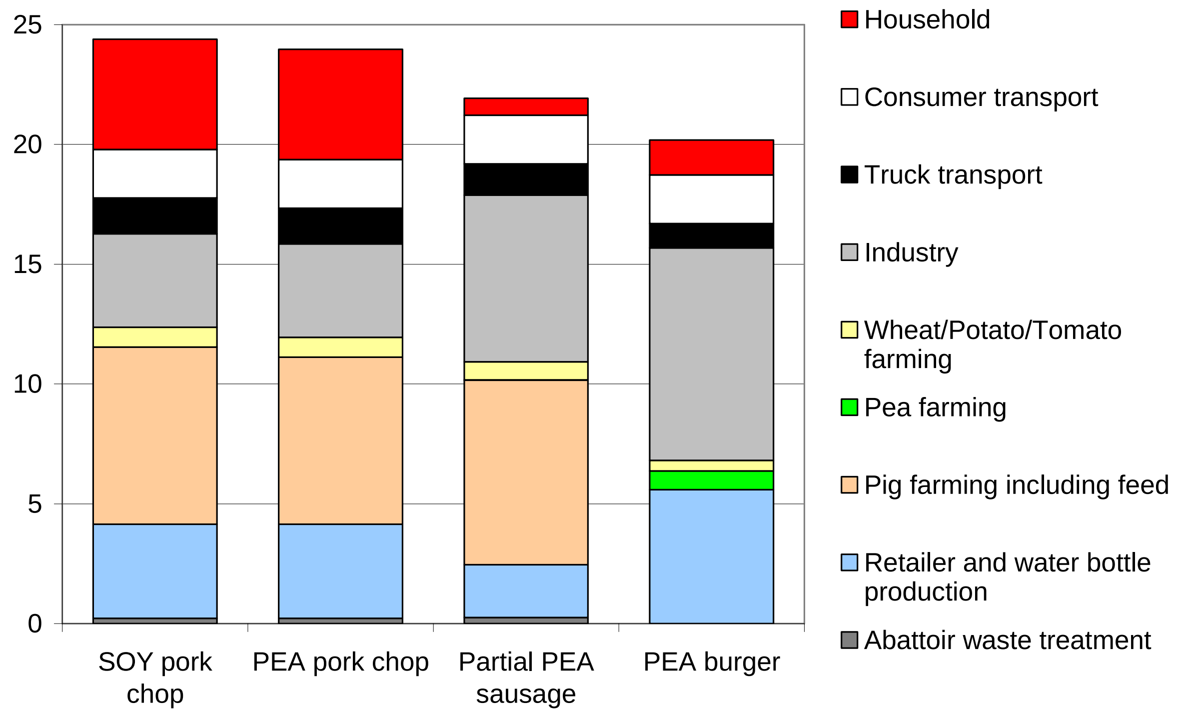

Fig. 2: Use of non-renewable and renewable energy for the Spanish meal scenarios [MJ-eq/meal]

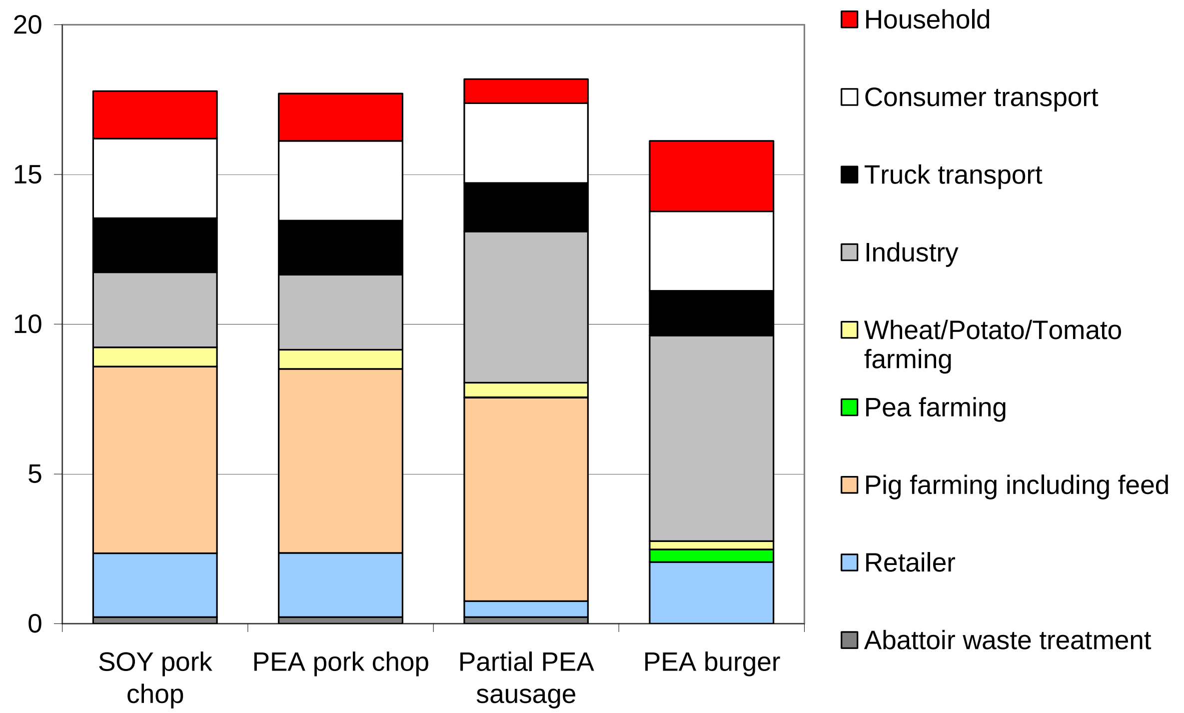

Fig. 3: Use of non-renewable and renewable energy for the Swedish meal scenarios [MJ-eq/meal]

Regarding contribution to global warming the meal with sausage has a higher contribution than the pork chop meals. This is a result of the fact that all meals must contain similar amounts of protein and energy; the amount of pork must be higher in this meal compared to the pork chop meals in order to fulfil these requirements. The pork chop meals contain a lot of potato in order to fulfil the recommended levels for energy content of a meal. The amount of sausage has to be as high as it is in the sausage meal to achieve the same level of protein as in the pork chop meals (which contain protein from both pork and potato). The contribution from production of peas for the pea protein in the sausage meal is negligible, so one way of decreasing the impact from the sausage meal would be to increase the share of pea protein in the sausage (which is only 10% of the total protein in the sausage in our case), but this was discarded for reasons of sensory quality. The vegetarian meal has a much lower contribution to global warming than the meals with animal protein (~50% lower for Swedish meals and 35% lower for Spanish). The reason for the smaller decrease in the Spanish case is due to the electricity production in Spain, which is partly based on combustion of coal. Since the pea burger meal requires a lot of electricity at the pea burger plant, retailer and at the household, the contribution is higher in the Spanish scenario compared to the Swedish (Swedish electricity production is largely based on nuclear and hydropower giving very low emissions of carbon dioxide), but the contribution is still only two thirds of that of the meals with animal protein.

The contributions to eutrophication for the four meals are shown in Fig. 4 and 5; here it is the production at the farm stage and the waste water treatment from the household that contribute. The total contribution from all other stages is relatively small. The level of protein is much the same in all four meals in each scenario, resulting in similar contributions from sewage treatment (included in ‘Household’). Overall, the contribution from the meals containing animal protein is a lot higher than the vegetarian meal. For the Swedish meals, there is very little difference between the two pork chop meals even though the feed compositions for the pigs are different. However, in the Spanish scenario, the contribution for the pork produced with pea based feed is higher than for the soy based pork. The reason for this is mainly due to nitrate leaching from the cultivation of peas. There is a higher incorporation of peas in the PEA formula in the Spanish scenario (18% of formula compared to 10% in the Swedish scenario), and the yield level for peas in Spain is comparatively low. The majority of the contribution from the pig farms comes from nitrate and ammonia (housing, manure spreading and piglet production).

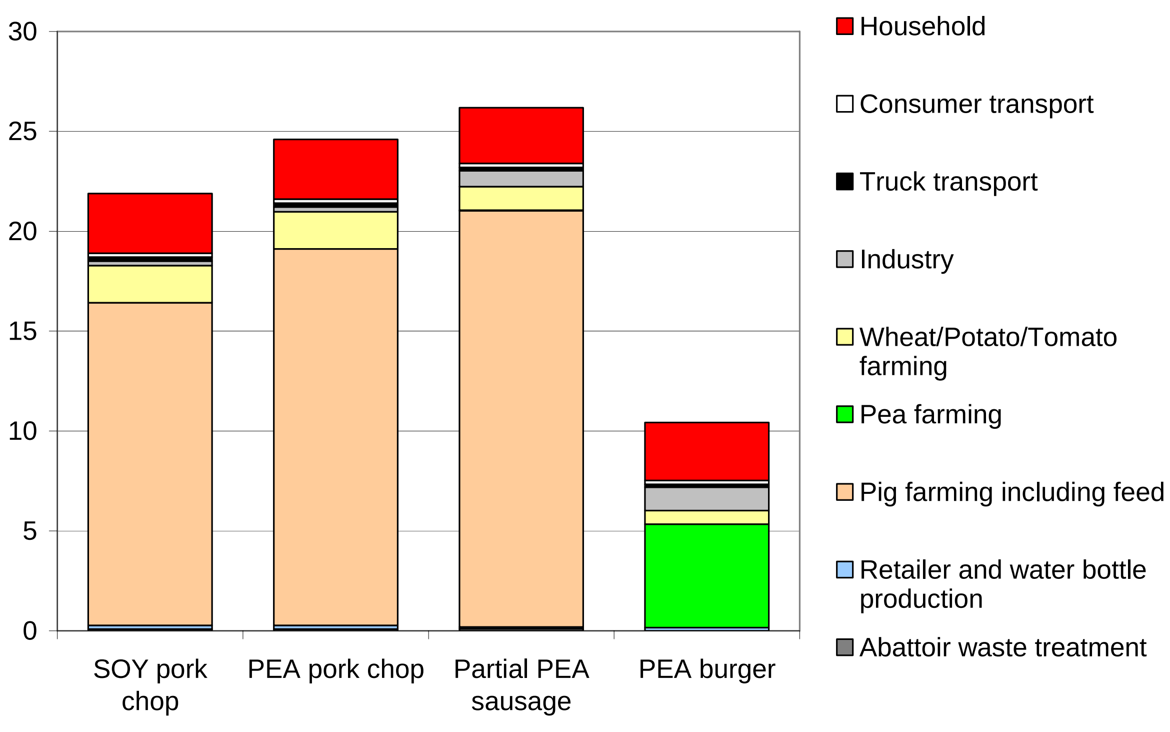

Fig. 4: Eutrophication potential for the Spanish meal scenarios [g N-eq/meal]

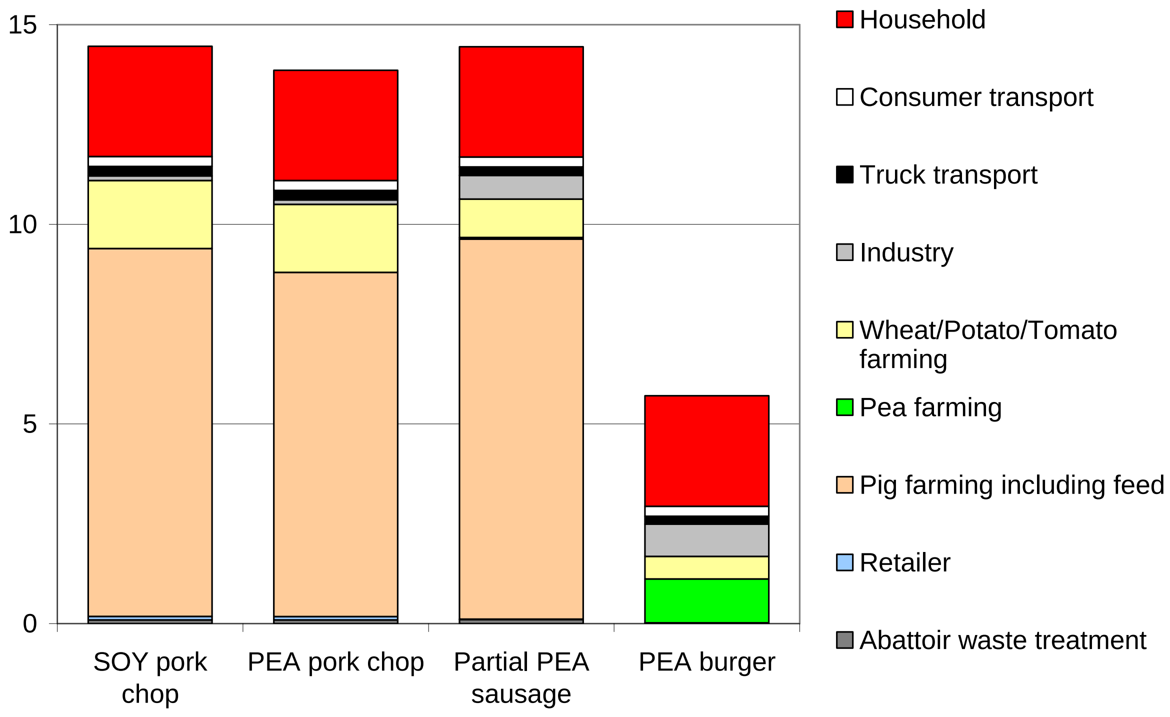

Fig. 5: Eutrophication potential for the Swedish meal scenarios [g N-eq/meal]

Discussion

The four meals that have been compared for each country's scenario are almost equal when it comes to the basic function of providing protein and energy. However, they are not the same if you consider other properties, e.g. they do not provide the same taste experience. Furthermore, nutrition can be broken down to more specific nutrients, e.g. proportion of essential amino acids, minerals and vitamins; nutrients which are *not* the same in the compared meals. This is a methodological issue within LCA, to compare the environmental impact of products that provide slightly different functions; in order to deal with this one must decide which function to give priority to, and in our case we have chosen the protein content and energy content.

In the analysis, the processing and storage of the pea burger proved to be an important contributor to the overall impact of the pea burger meal. The data used for the processing comes from a small scale producer of vegetarian products. It is likely that another plant with a larger production volume would result in a lower environmental impact per produced unit. Furthermore, only one plant was inventoried; further information from other production facilities is therefore needed to validate the data used in this analysis.

Conclusion

In summary, the environmental benefit of wholly pea based protein in a meal is clear, but there is scope for improving the energy efficiency in the processing and storing of frozen vegetarian products. Moreover, the study shows it is important to look at a complete meal, and to follow the entire processing chain up to consumption, when comparing the environmental impact of different protein sources.

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Accounting for biogenic NMVOC emissions in LCA

Jungbluth, Niels
ESU-services Ltd. fair consulting in sustainability, Kanzleistrasse 4, CH-8610 Uster, T. +41 44 940 6132, F. +41 44 940 61 94, jungbluth@esu-services.ch

Keywords: isoprene, monoterpene, terpenes, NMVOC, photochemical oxidation, summer smog, ozone formation, vegetation emissions

Abstract

Isoprene is emitted in vast amounts from photosynthesizing leaves of many plant species, particularly by trees. They are a major contributor to the total biogenic volatile organic compound (BVOC) flux. The substance plays an important role in atmospheric chemistry. Isoprene rapidly reacts with hydroxyl radicals in the atmosphere. In the presence of nitric oxides (NO_X), the oxidation of isoprene contributes to the formation of ozone. Moreover, isoprene also contributes to the regulation of tropospheric hydroxyl radical concentration and thus plays an important role in determining the abundance of atmospheric methane, an important greenhouse gas.

So far, such biogenic NMVOC (non-methane volatile organic compounds) emissions are only rarely accounted for in LCA of biomass products. There is a modelling uncertainty in LCI due to several influencing factors like type of plant, temperature or irradiation of the sun. In addition, there is a large seasonal variation with the main emissions soon after bud break in summer and quite lower emissions in winter.

A case study has been conducted for producing and using BTL-fuels (biomass-to-liquid) from straw, miscanthus, and short-rotation wood. This conference paper focuses on the results for category indicators characterising NMVOC emissions, e.g. ozone formation. NMVOC of plants have a large effect on the total environmental impacts in the life cycle of products from renewable resources if accounted for.

Pros and Cons of including such emissions in LCA studies are discussed in the end of this paper. It is debatable whether such emissions should be included because they also arise from non-cultivated biomass areas. Thus, the conclusion of taking them into account would be to reduce the area actually covered by biomass.

Introduction

"Isoprene (also known as 2-methyl-1,3-butadiene), an unsaturated C-5 hydrocarbon, is emitted in vast amounts from photosynthesizing leaves of many plant species, particularly trees. With a global atmospheric carbon flux of approximately 450 million tons of carbon per year, isoprene emissions are a major contributor to the total biogenic volatile organic compound (BVOC) flux of 1,200 million tons of carbon per year. This is in the same order of magnitude as anthropogenic emissions. Current interest in understanding the biochemical and physiological mechanisms controlling isoprene formation in plants comes from the important role isoprene plays in atmospheric chemistry. Isoprene rapidly reacts with hydroxyl radicals in the atmosphere. In the presence of nitric oxides (NO_X), the oxidation of isoprene contributes significantly to the formation of ozone, a dominant tropospheric air pollutant. Moreover, isoprene also contributes to the regulation of tropospheric hydroxyl radical concentration and thus plays an important role in determining the abundance of atmospheric methane, an important greenhouse gas." On a sunny day the isoprene emission of 10,000 trees can be up to 10 kilograms per hour.[1]

[1] Information from http://www.plantphysiol.org/cgi/content/full/135/1/152 retrieved on 11.2005.

Method / Approach

So far biogenic NMVOC emissions are only rarely accounted for in LCA. There is a modelling uncertainty due to several influencing factors like type of plant, temperature or irradiation of the sun. Also it has been shown that there is a large seasonal variation with the main emissions soon after budbreak in summer and quite lower emissions in winter. No information could be found about the influence of different cultivation intensities (e.g. fields with lower or higher annual yields). Nevertheless, according to the today's knowledge, these emissions are quite important with respect to the formation of summer smog and thus they should be accounted for in LCI.

The difficulties with estimating such emissions are visible from showing some results for the annual emissions per hectare. Tab. 1 shows an overview of results from selected studies that vary by several orders of magnitude.

Tab. 1: Estimation of NMVOC emissions in different studies (kg/ha/year)

Pollutant	Plant	Range	Mean	Reference
Isoprene	Poplar	189-1600	476	(Mann & Spath 1997)
Monoterpene	Swiss forest	Factor 5	29	(Spirig & Neftel 2002)
VOC	Swiss agriculture	-	4	(Spirig & Neftel 2002)
VOC	Swiss grasslands	-	3.6	(Spirig & Neftel 2002)
NMVOC	German area	5-25	-	(UMEG 2000)

The NMVOC emissions during plantation of Straw, Miscanthus and Short-rotation wood have been investigated for a life cycle assessment of producing BTL-fuels (biomass-to-liquid) from these types of biomass (Jungbluth *et al*. 2007c; Jungbluth & Schmutz 2007). These type of biofuels are also sometimes referred to as "second generation" fuels because in a new type of process several different types of ligno-cellulosic biomass can be used. Results from this case study are presented in this paper.

Life cycle inventory analysis

The emission rates of plant species are normally measured as microgram of isoprene emission per hour and gram of dry matter leaves under standardized temperature and irradiation conditions. This factor is multiplied with the leaf mass and a correction factor accounting for the regional available amount of sunlight. Tab. 2 provides the estimation used in this study based on the model of Richardson (2002:page B1101-1-19). This model allows accounting for regional differences in Europe and plant specific factors.

Leaf weight (kg/ha) and emission factors for miscanthus and wheat (kg/kg leaf/h) are estimated based on Sanderson (2002). The amount of harvested biomass is taken from the inventories of biomass production (Jungbluth *et al*. 2007c; Jungbluth & Schmutz 2007). The leaf weights are only available as averages for different types of biomass and thus do not account for different amounts of harvest. This has been corrected by multiplying the emission factor with the actual harvest divided by the average harvest of these cultures. An "environmental correction factor" accounts for the differences e.g. in irradiation, sunshine hours or temperature (Sanderson 2002). The factors for different countries are shown in the report (Jungbluth *et al*. 2007c; Jungbluth & Schmutz 2007).

The general difference between emissions from forests and agriculture is known and thus the higher amount of emissions from willow-salix compared to agricultural products can be assumed to be correct. In contrast, the difference between wheat and miscanthus is too small and considered as insignificant.

The considered time period takes into account a full cultivation period for perennial crops. Tab. 2 shows that the average amount of emissions per hectare and year is about 20 to 50 kg. These figures are in the order of magnitude of other publications as shown in Tab. 1. The overall uncertainty is estimated with 5 according to the ecoinvent methodology. The emissions for wheat growing have been allocated between straw and grains based on an economic allocation. Thus, straw bears relatively low emissions per kg of dry matter.

Tab. 2: Emission rate for isoprene and monoterpene emissions used in this study.

	leaf weight (kg/ha)	biomass harvest (kg dry matter/ha/period)	Isoprene (kg/kg leaf/h)	other NMVOC (kg/kg leaf/h)	Isoprene (kg/ha/a)	Monoterpene (kg/ha/a)
Willow-Salix	1500	176'844	3.40E-05	1.70E-06	53.1	2.7
Miscanthus	1250	15'547	1.60E-05	8.00E-07	21.6	1.1
Wheat	1250	8'618	1.60E-05	8.00E-07	20.1	1.0

Results

Introduction

In the base case, biogenic NMVOC emissions are excluded from the assessment by setting the characterisation factor of “isoprene, low population area” to zero in this adapted method. There is no characterisation factor given by Guinée *et al.* (2001a) for (mono-)terpene, the other NMVOC emission investigated in the LCI for biomass production. A sensitivity analysis was performed, considering the isoprene emissions.

The methodology EDIP 2003 is used here for a sensitivity analysis as this can be used to characterize all NMVOCs including isoprene and terpenes (Hauschild & Potting 2005).

Ozone formation of biomass production

Fig. 1 evaluates the results for ozone formation due to the production of three different types of biomass. Two slightly different scenarios are calculated for the biomass production. Due to differences in emissions there are large differences between the different types of biomass. The biogenic isoprene and terpene emissions make an important contribution to the total LCIA results.

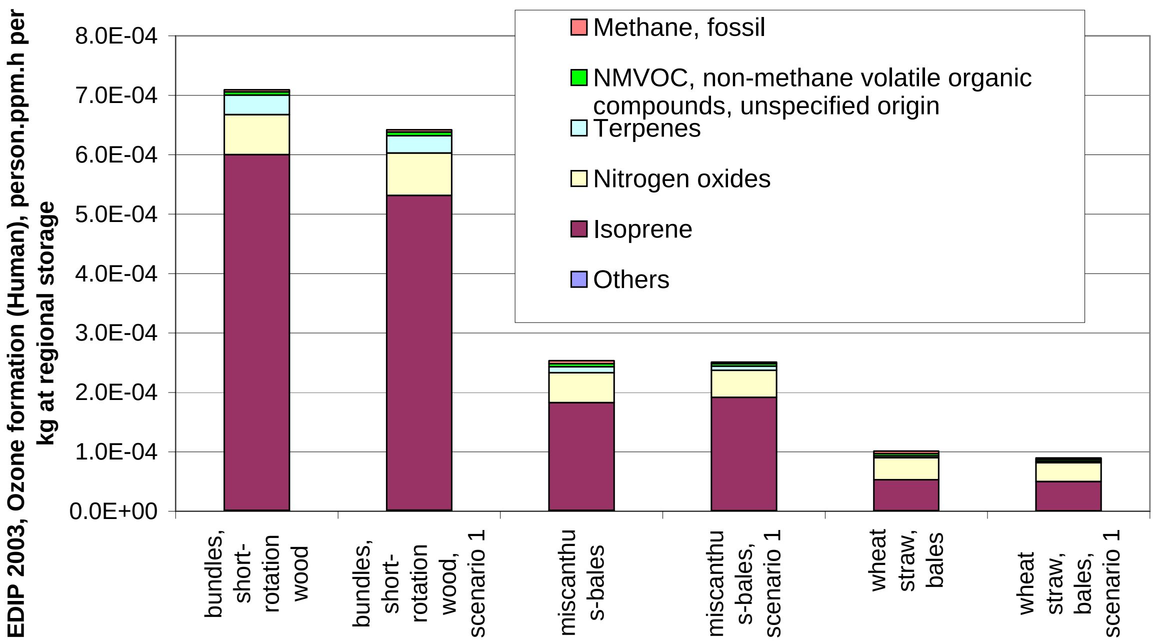

Fig. 1: Contribution of individual elementary flows to the total EDIP 2003, Ozone formation (Human), person.ppm.h per kg of biomass at regional storage

Production of BTL-fuels

Fig. 2 shows the results for comparing different fuel production processes including biogenic NMVOC emissions with EDIP 2003 (Hauschild & Potting 2005). The air emission of pollutants contributing to ozone formation is dominated by the biomass production. The conversion ratio in fuel production and the type of biomass use are quite important. Only for processes based on straw, other types of emissions get some relevance because of the much lower isoprene emissions allocated to straw.

Processes based on straw or miscanthus, have a clear advantage in comparison to processes based on wood with regard to this category indicator. This should be taken into account in process development, even if the inventory of these substances might still have an uncertainty of about factor 2.

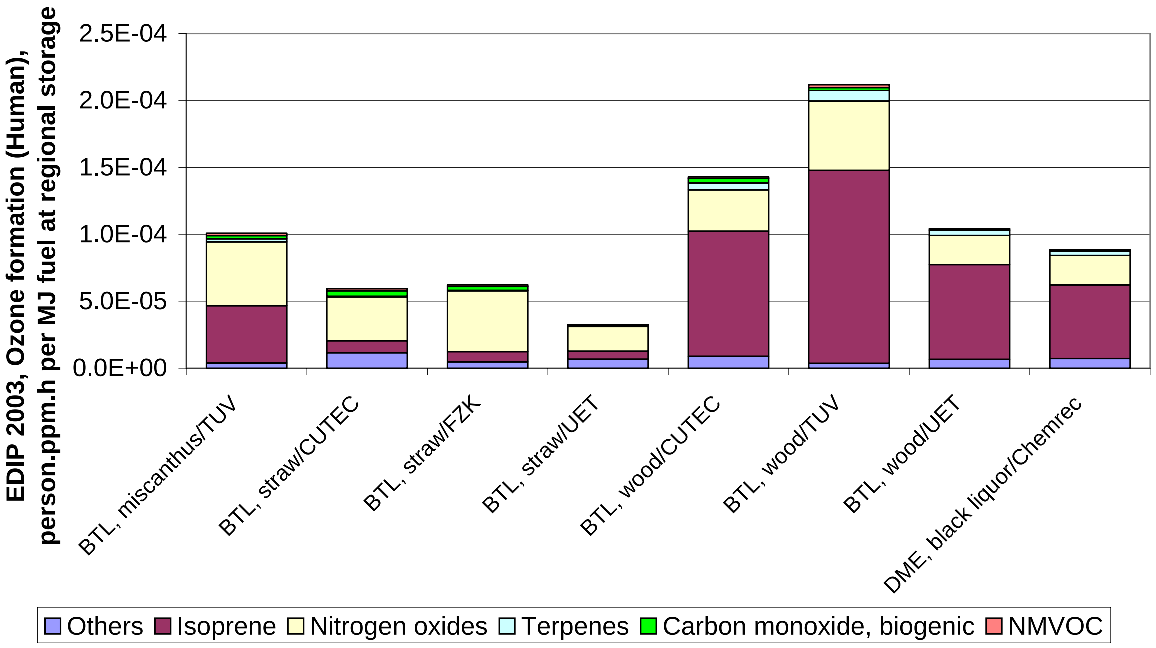

Fig. 2: Comparison of different BTL-fuel production processes with the category indicator ozone formation (human) according to the EDIP 2003 methodology

Other LCIA methods for photochemical smog

A detailed analysis for the biomass production showed that emissions of SO_x and CO are important with regard to non-biogenic photo-oxidant formation if the CML indicator is used. They are emitted in several different processes in the life cycle. An important input is the use of tractors, which includes the emissions from the supply of the fuel and from producing the tractor.

A sensitivity analysis with the CML 2001 method, including the characterisation of biogenic NMVOC emissions has been made. Isoprene emissions are by far the most dominant emissions accounting for about 99 % of the cumulative photochemical oxidation potential if they are included in the assessment. For the indicator photo-oxidant formation there are advantages for the use of straw and miscanthus that emit lower amounts during growing.

In Fig. 3 we perform a sensitivity analysis of the category indicator photochemical smog with the older EDIP 97 methodology (Hauschild & Wenzel 1997). Isoprene emissions from biomass production are dominant. Unspecified NMVOC, which are not accounted for in the CML methodology, are important for the processes based on straw input. On the other side sulphur dioxide is not accounted for by this method. The ranking of the different processes is not much influenced by the choice of LCIA method and the exclusion or inclusion of some individual emissions. But, all methods for ozone formation show a dominance of biogenic NMVOC emissions if included in the assessment.

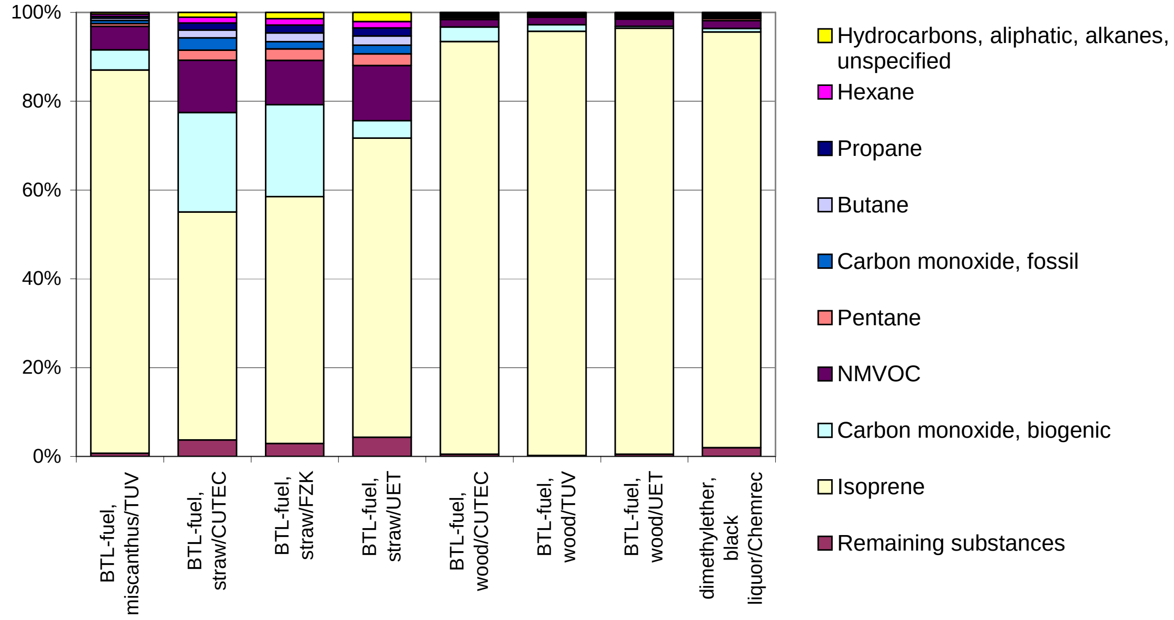

Fig. 3: Contribution of individual elementary flows to the total photochemical smog, EDIP 97 methodology, for BTL-fuel production

Using BTL-fuels in cars

The use of BTL-fuels and a comparison with fossil fuels have been investigated in a follow up study (Jungbluth *et al*. 2008). Biogenic NMVOC emissions have been excluded here for consistency reasons with former studies in the same project (Jungbluth *et al*. 2007a; Zah *et al*. 2007). The environmental impacts regarding ozone formation are analyzed in Fig. 4 for the fuel use in passenger cars. The EDIP 2003 method has been used with the characterisation factors for ozone formation (human) (Hauschild & Potting 2005).

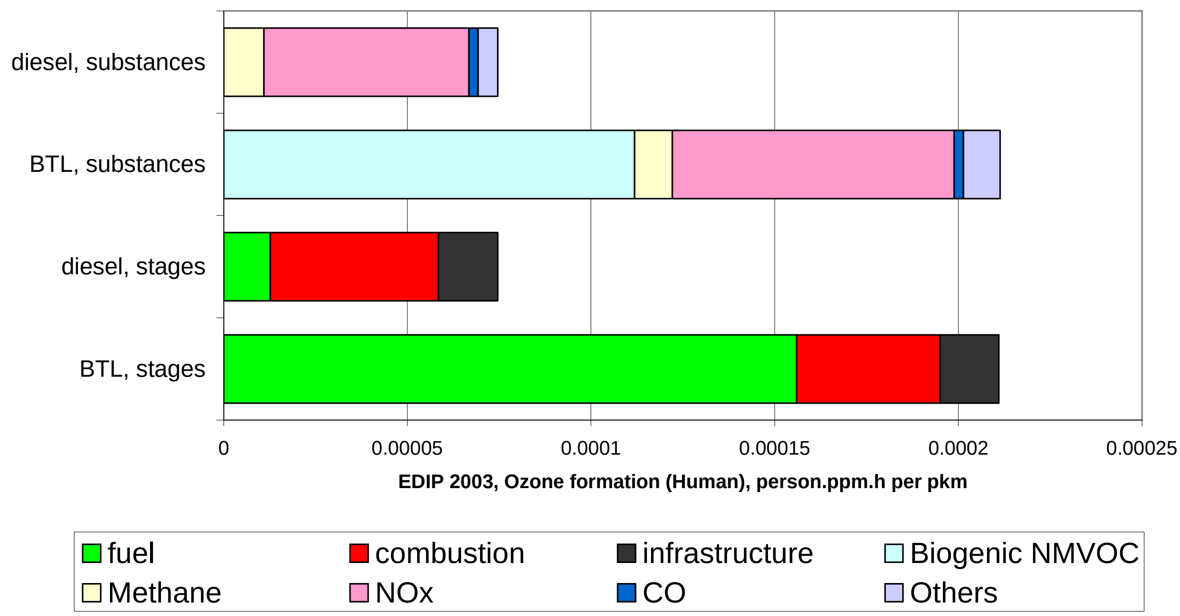

Fig. 4: Comparison of using fossil diesel and BTL-fuel from short-rotation wood with regard to ozone formation

Compared to diesel, using BTL in a passenger car causes lower tailpipe emissions of air pollutants contributing to the problem of ozone formation (combustion in Fig. 4). But emissions during fuel production from short-rotation wood are much higher. This can also be seen if the contributing substances are analyzed. For BTL fuel biogenic NMVOC emissions are quite important, while other emissions are about the same over the full life cycle compared to diesel fuel.

Conclusions

The following conclusions can be drawn for the importance of biogenic NMVOC emissions in the life cycle assessment of renewable resources. NMVOC emissions from growing plants contribute substantially to the photochemical smog indicator. This can be found with several different LCIA methodologies for this problem if the specific type of emissions has a characterisation factor in the method. The dominance is still true if the full life cycle of a renewable product, e.g. a biofuel is investigated. The surplus emissions can also outweigh other improvements in the life cycle e.g. lower tailpipe emissions in the case of BTL-fuels.

Still, there are some uncertainties concerning differences between species, regions, natural conditions, etc.. Thus, it is necessary to establish a good database covering all type of plants and regions in a comparative manner.

Biomass resources with low NMVOC emissions should be a criterion in LCA of renewable resources or products made from renewables. An important aspect might be the reduction of NMVOC emissions from plants by choosing favourable types of biomass resources. Grassland and agriculture are in general more favourable than forest biomass regarding this aspect.

Discussion

In any case, it has to be taken into account that the formation of summer smog depends not only on the amount of NMVOC in the atmosphere, but also on other pollutants, e.g. NO_x and on the presence of sun light. Thus, it is quite difficult to model a linear relationship in the LCIA between the NMVOC in agricultural areas and the formation of summer smog. On the one side it can be assumed that in rural areas NO_x is the limiting factor for formation of summer smog, but on the other side it is known that in rural areas ozone formation might be higher due to missing reactants for the degradation of ozone.

A critical issue is the inclusion of biogenic NMVOC emissions while comparing renewables with conventional products. In many cases such “bio”-products will show higher emissions contributing to ozone formation than products made e.g. from fossil resources. Thus, the conclusion would be to grow as little biomass as possible. But, this does not make sense as large parts of agricultural or forestry land will be covered with biomass regardless weather it is used for products or not.

Thus, it might be an option to take into account only those emissions that are surplus compared to a natural state of the land area. For determining such a reference state it would be necessary to know what would be grown on the land area if no biomass production for the investigated product would occur. In most cases there would also be some type of biomass growing. Thus, only the net balance between the reference state and the actual production patterns can be included in the modelling of NMVOC emissions. Thus much lower or even negative emissions would be accounted for.

Such a discussion would be similar to the discussion for determining the CO_2 emissions due to land use changes.

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Method for considering life cycle thinking and watershed vulnerability analysis in the environmental performance evaluation of agro-industrial innovations (Ambitec-Life Cycle)

M. C. B. Figueirêdo[1], F. S. B. Mota[2], G. S. Rodrigues[1], A. Caldeira-Pires[3], M. F. Rosa[1] and V. P. P. B. Vieira[2]

[1]Brazilian Agricultural Research Corporation – Embrapa, Fortaleza (CE) and Jaguariúna (SP), Brazil, clea@cnpat.embrapa.br; [2]Federal University of Ceará, Fortaleza (CE), Brazil; [3] University of Brasília, Brasília (DF), Brazil

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Abstract

The current emphasis on sustainable development warrants the development and adoption of innovations to render industrial production more efficient in the use of natural resources and less polluting. In order to develop innovations for sustainability, management models and evaluation tools must integrate objective environmental considerations. One such tool is the Ambitec-Agro System, a set of integrated indicators specifically proposed to assess environmental impacts of agro-industrial innovations. This System compares an innovation's environmental performance against the pre-existing technology, focusing the analysis on the innovation-adopting establishment scale. This study presents a conceptual method that expands the scope of Ambitec-Agro by including life cycle thinking and watershed vulnerability analysis to the environmental performance evaluation of agro-industrial innovations. In order to develop this approach, the steps inherent to a multi-criteria decision support system were followed. The proposed method includes four life cycle phases to evaluate the environmental performance of an agro-industrial innovation: (i) raw material production used by innovation, (ii) innovation production, (iii) innovation use and (iv) its final disposal. The method also includes a vulnerability analysis of the watersheds where each life cycle phase takes place. The proposed integrated method provides decision makers a broadened view of an agro-industrial innovation environmental performance, shedding light on technological improvements throughout its entire life cycle.

Introduction

According to the World Business Council for Sustainable Development (WBCSD), environmental sustainability requires the development of innovations that contribute to the efficient use of natural resources (WBCSD, 2001). In consonance with this directive, the Ambitec-Agro System (Rodrigues *et al*., 2003) has been used since 2001 to assess the environmental impacts of agro-industrial innovations proposed by research and development (R&D) programs carried out at the Brazilian Agricultural Research Agency (Embrapa). This System integrates environmental impact indicators in weighing matrices designed to compare the performance of a given innovation with the performance of a previously existing technology, focusing the analysis on the productive unit (the rural establishment or agroindustry) where the innovation is adopted (Monteiro & Rodrigues, 2006).

However, during the last decade, the scientific community witnessed the intensification of the debate about the importance of evaluating the impacts of products or services along their production, consumption and post-consumption phases, that is, along their life cycle. The Society of Environmental Chemistry and Toxicology (SETAC) and other institutions have sponsored workshops and projects to develop a conceptual framework for conducting life cycle assessments (LCA). This framework is formally presented in the ISO 14040 and 14044 standards (Roy *et al*., 2009).

LCA of agro-industrial products is spreading with the development of impact assessment methods that consider emissions from the use of agrochemicals and their impacts on the environment (Roy *et al*., 2009; Nemecek *et al*., 2008). However, some difficulties still contribute to the restricted use of LCA in certain countries such as Brazil: the scarcity of locally detailed databases to support data inventories, despite recent efforts such as the first Brazilian database on energy production in 2007 (Ferreira *et al*., 2007); the lack of consolidated methods to evaluate impacts on soil, such as erosion, salinization and compaction, and impacts on water availability, all issues of special interest to the Brazilian context, especially in semi-arid areas (Pennington *et al*., 2004; Pegoraro, 2007).

The consideration of the environmental vulnerability of a natural system that receives emissions released in a life cycle phase is also important, since each system is affected differently depending on its socioeconomic and environmental characteristics. Although the vulnerability concept is not consensual in scientific terms, according to Adger (2006), it is usually linked to one or more of the following factors: exposure, system's sensitivity and adaptive capacity. Exposure means the level, duration or extension of the system contact with perturbations. Sensitivity is related to the system's ecological capacity to assimilate environmental pressures without being degraded in the long run. System's adaptive capacity concerns the ability to make use of resources or respond to pressures, preventing, controlling or remediating environmental degradation. The quantification of these factors allows the evaluation of a system's vulnerability to specific environmental pressures, with a system being more vulnerable when exposure and sensitivity are high and adaptive response is low.

The LCA framework according with ISO 14040 does not considerer a system vulnerability to consumptions and emissions related to a studied product life cycle. Nonetheless, some life cycle impact assessment methods such as EDIP (Potting & Hauschild, 2005) and TRACI (Bare *et al*., 2003) developed site-dependent characterization factors to consider spatial differentiation in some impact categories, at a regional level. The consideration of the characteristics of the surrounding environment is especially important in the impact assessment of agricultural activities.

This study presents a conceptual method named Ambitec-Life Cycle that considers life cycle reasoning and watershed vulnerability analysis in the environmental performance evaluation of agro-industrial technological innovations. The proposed method aims to subsidize agro-industrial innovations' R&D, showing critical points in an innovation life cycle that need to be addressed to innovations reach better environmental performance than its substitute technology. This method is based on and expands the scope of the Ambitec-Agro System.

Method

In order to develop Ambitec-Life Cycle, the steps described below were followed, as proposed by Malczewski (1999) for the delineation of a multi-criteria decision support system:

(i) <u>Definition of the decision question to be addressed</u>: the decision question is: how to expand the Ambitec-Agro System to consider different phases of an innovation's life cycle and the vulnerability of the environment where each phase of its life cycle occurs?

To answer this question, it is necessary to make it clear what is understood by life cycle and vulnerability. The life cycle concept adopted is present in the ISO 14040 standards: life cycle is related to successive and connected stages of a production system, from raw material acquisition to product final disposal. The vulnerability concept adopted is based on Adger (2006), applied to the watershed scale and encompasses: exposure to human pressures that have the potential to cause environmental impacts; sensitivity of the ecological system to the pressures; and local society capacity of response to the environmental pressures.

The following environmental impacts were pointed out as relevant to the study of agro-industrial activities: (i) loss of biodiversity, (ii) soil erosion, (iii) compaction, (iv) salinization, (v) sodification, (vi) acidification, (vii) desertification, (viii) environmental contamination by agrochemicals and (ix) solid wastes, (x) water scarcity and (xi) pollution, (xii) depletion of non-renewable resources, (xiii) climate change and (xiv) food contamination by use of additives (Figueirêdo, 2008).

ii) Identification of possibilities to apply the multi-criteria analysis: as in the Ambitec-Agro System, the analysis is applied to two technologies - the focused innovation and a substitute technology already being used with a similar function in the market. By comparing the environmental performance of an innovation with the performance of its substitute technology, it is possible to identify whether the innovation causes more or less impact than its substitute technology and to proceed with changes and improvements in the innovation characteristics, if necessary.

iii) Definition and organization of indicators and indices: the hierarchical multi-criteria structure of the Ambitec-Agro System (Rodrigues *et al*., 2003) was expanded to consider other life cycle phases of the studied innovation and its substitute technology. This multi-criteria structure organizes environmental indicators in principles and criteria, aggregated as an environmental performance index.

To perform the vulnerability analysis, a multi-criteria structure that organizes indicators in criteria and in a watershed vulnerability index was also developed.

iv) Definition of rules to the multi-criteria analysis: the rules established standards to process data in the proposed method and were based on the multi-criteria theory revised by Malczewski (1999).

v) Sensitivity analysis: with the quantitative methods in place, simulations were performed with each indicator assuming variations (±10%, ±50%, change to zero and change from zero to a greater number), in order to measure the sensitivity of the method, as proposed by Jorgensen (1994).

Results

The conceptual method of Ambitec-Life Cycle and the main steps necessary to its implementation are shown in Fig. 1. Four life cycle phases are considered for a given innovation, instead of just the use phase as in the scope of the original Ambitec-Agro System: raw material production used by the innovation, innovation production, innovation product use and its final disposal.

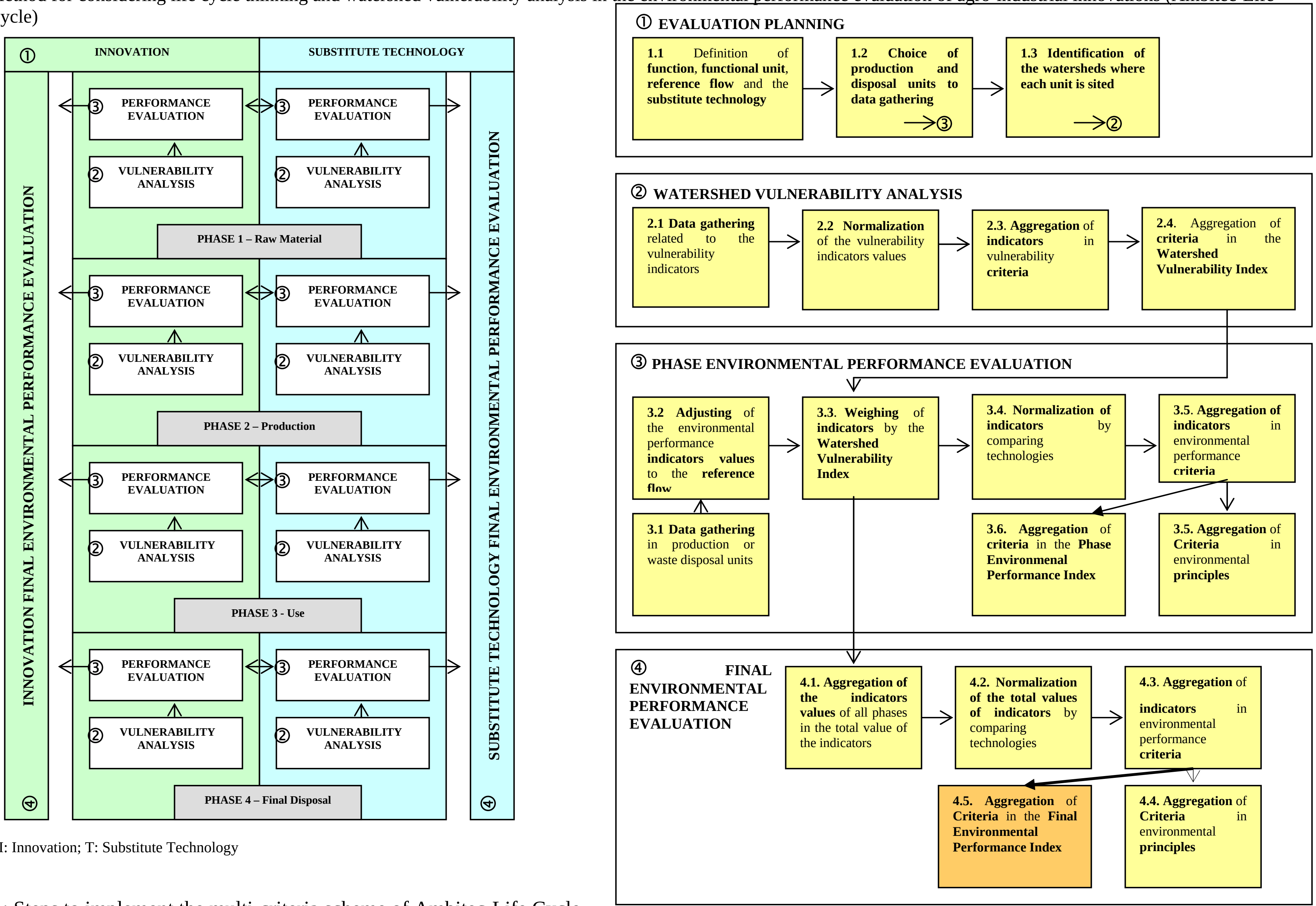
If an innovation uses a byproduct or residue as a raw material, the first phase is not considered. However, if the use of such an innovation leads to the disposal of byproducts or residues formerly used by its substitute technology, this disposal must be accounted for in the raw-material phase.

The environmental performance analysis along these life cycle phases must be carried out to an innovation and to its substitute technology, available in the market. The multi-criteria structure containing the set of principles, criteria and quantitative indicators chosen to assess the environmental performance of an innovation and its substitute technology are presented in Fig. 2. The set of indicators are related to environmental issues that concern agriculture, agro-industry or final disposal activities. Some of them are more relevant to one activity while others can be used by anyone of the aforementioned activities.

As each phase of the innovation and of its substitute technology can take place in different watersheds, the environmental vulnerability analysis is performed for each concerned one. The vulnerability analysis is based on a multi-criteria scheme that links environmental indicators to criteria and to a watershed Environmental Vulnerability Index (EVI) (Fig. 3).

The EVI enters the performance evaluation of an innovation and of its substitute technology as a weight to those indicators that represent consumptions and emissions with potential to cause environmental impacts in the watershed area. These indicators are shown in Fig. 2. The higher the vulnerability of a watershed, the higher the potential effect of indicators related to environmental issues of relevant importance at the watershed level. This procedure highlights those consumptions and emissions of an innovation or its substitute technology that can lead to environmental impacts at the watershed level, when the watershed vulnerability is high.

The results of the analysis of each life cycle phase are aggregated to obtain a concluding environmental performance evaluation of an innovation and its substitute technology.



Where: I: Innovation; T: Substitute Technology

Fig. 1: Steps to implement the multi-criteria scheme of Ambitec-Life Cycle

ENVIRONMENTAL ISSUES

- Loss of biodiversity
- Soil erosion
- Soil compaction
- Soil salinization and sodification
- Soil acidification
- Agrochemicals environmental contamination
- Solid waste environmental contamination
- Desertification
- Water Scarcity
- Water pollution
- Climate Change
- Depletion of non-renewable material and energy sources
- Food contamination by additives

INDICATORS	CRITERIA
1.1 Total quantity of materials (G, D*) 1.2 Total quantity of dangerous materials (G, D*) 1.3 Total quantity of non-renewable materials (G, D*) 1.4 Total quantity of not recycled/reused materials (G, D*)	1. Consumption of materials
2.1 Total quantity of electricity (G, D*)	2. Consumption of Electricity
3.1 Total quantity of fuels (G, D*) 3.2 Total quantity of fossil fuels (G, D*) 3.3 Total quantity of fuels not obtained from waste (G, D*)	3. Consumption of fuel
4.1 Total water volume (W, G, D)* 4.2 Total water volume not recycled/reused (W, G, D*)	4. Consumption of water
5.1 Deforestation area (W, A, D*) 5.2 Degraded area recovered (W, A, D*)	5. Vegetation management
6.1 Quantity of macronutrients (W, A*) 6.2 Quantity of micronutrients (W, A*)	6. Consumption of fertilizers
7.1 Quantity of pesticides (W, A*)	7. Consumption of pesticides
8.1 Product lifetime (A, G)	8. Product durability
9.1 Risk class of genetically modified organism (GMO) (W, A*)	9. Use of GMO
10.1 Food addictive total limit (AI)	10. Use of food addictives
11.1 Total quantity of waste (W, G*) 11.2 Total quantity of dangerous waste (W, G*) 11.3 Total quantity of not recyclable or reusable waste (W, G*)	11. Solid waste generation
12.1 Area of exposed soil (W, A*) 12.2 Area of mechanized soil (W, A*)	12. Soil erosion and compaction
13.1 Salinity of irrigation water (W, A*) 13.2 Sodicity of irrigation water (W, A)*	13. Irrigation water quality
14.1 Burned agriculture area (A*) 14.2 Total quantity of burned waste (G D*)	14. Waste burning
15.1 Biochemical Oxygen Demand load (W, G, D*) 15.2 Chemical Oxygen Demand load (W, G, D*) 15.3 Total Suspended Solid load (W, G, D*) 15.4 Total Kjeldahl Nitrogen load (W, G, D*) 15.5 Total Phosphorous load (W, G, D*) 15.6 Total Oil and Grease load (W, G, D*) 15.7 Electric Conductivity (W, G, D*) 15.8 Volume of the effluent not recycled/reused (W, G, D*)	15. Effluents generation
16.1 Quantity of organic waste sent to landfill (W, D)*	16. Organic waste anaerobic decomposition
17.1 Flooded irrigation area	17. Flood irrigation

PRINCIPLES

- TECHNOLOGY EFFICIENCY (Criteria 1, 2, 3, 4, 5, 6 and 7)
- BIODIVERSITY CONSERVATION (Criteria 5, 7 and 9)
- SOIL CONSERVATION (Criteria 6, 7, 11, 12, 13 and 14)
- WATER CONSERVATION (Criteria 4, 6, 7, 15 and 16)
- AIR CONSERVATION (Criteria 3, 6, 7, 14, 16 and 17)
- PRODUCT QUALITY (Criteria 8 and 10)

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ENVIRONMENTAL PERFORMANCE INDEX

* W - Environmental performance indicators that are weighed by a watershed environmental vulnerability index (EVI); A – indicators related to agriculture; AI – indicators related to agro-industry; G – indicators related to agro-industry and agriculture; D – indicators related to final disposal.

Fig. 2: Set of environmental performance indicators, criteria and principles available to the environmental performance evaluation of a technology

ENVIRONMENTAL ISSUES
Loss of biodiversity
Soil erosion
Soil compaction
Soil Salinization/ sodification
Soil acidification
Pesticide environmental contamination
Solid waste environmental contamination
Desertification
Water scarcity
Water pollution

INDICATORS	CRITERIA
1.1 Agriculture activity	1. EXPOSITION
1.2 Industrial activity	
1.3 Generation of wastewater *per person*	
1.4 Generation of waste *per person*	
1.5 Water demand *per person*	
2.1 Priority areas for conservation	2. SENSITIVITY
2.2 Agriculture capability	
2.3 Rainfall intensity	
2.4 Irrigation water quality	
2.5 Climate aridity	
3.1 Areas in conservation units	3. CAPACITY OF RESPONSE
3.2 Soil conservation	
3.3 Access to treated water	
3.4 Access to waste collection and appropriate disposal	
3.5 Access to sanitation	
3.6 Water availability *per person*	
3.7 Municipality Human Development Index (HDI-M)	

INDEX
ENVIRONMENTAL VULNERABILITY INDEX

Fig. 3: Set of indicators and criteria to perform a watershed environmental vulnerability analysis

The main steps necessary to implement the Ambitec-Life Cycle method are:

i) Evaluation planning

The planning step of an innovation environmental performance evaluation begins with the definition of its function, functional unit, substitute technology and the reference flow. A function of an innovation is defined looking at its purpose when adopted. The functional unit is a quantification of an innovation (process or product) function. A substitute technology is chosen because it has a function similar to that of an innovation. The reference flow is the measure of intermediate and final products necessary to fulfil an established functional unit. An innovation and its substitute technology have a common function and functional unit and specific reference flows.

The next step is the choice of the production and disposal units for data collection on the environmental performance indicators. Finally, the watersheds where each unit is located are identified.

ii) Watershed vulnerability analysis

To carry out the vulnerability analysis, it is first necessary to gather data related to the set of vulnerability indicators.

Because each indicator has a different measuring unit, they must be normalized to a common dimensionless scale in order to allow their aggregation in criteria and in a watershed EVI. This index enters as a weighing factor in the environmental performance evaluation of an innovation and its substitute technology, in a given phase of their life cycle.

Vulnerability indicators can be quantitative (e.g. water demand and availability) or qualitative (e.g. agriculture capability and climate aridity). The “score range” rule, proposed by Malczewski (1999) is used to normalize the quantitative indicators of environmental vulnerability. This rule converts an indicator value to a standardized score in a scale ranging from 1 to 2, where 1 represents the lowest vulnerability and 2, the highest. The maximum and minimum values are obtained from literature and from available national databases.

Quantitative indicators in the proposed method belong to one of two groups: “the higher their value, the higher the environmental vulnerability” and “the higher their value, the lower the vulnerability”. Formulas 1a and 1b are used to normalize indicators that belong to the first and second group, respectively.

$$Value_i = \left(\frac{indicator_i - Value_{\min}}{Value_{\max} - Value_{\min}} \right) + 1 \qquad \text{(Formula 1a)}$$

$$Value_i = \left(\frac{Value_{\max} - indicator_i}{Value_{\max} - Value_{\min}} \right) + 1 \qquad \text{(Formula 1b)}$$

In Formulas 1a and 1b, “Indicator_i” represents the measured value of vulnerability indicator *i*; “$\text{Value}_{\max}$” is the maximum value that indicator *i* can assume; “$\text{Value}_{\min}$” is the minimum value that indicator *i* can assume and “Value_i” is the normalized value of indicator *i*.

For the qualitative indicators, a score is attributed to each possible response, ranging from 1 to 2, according to the understanding of the situation representing lower or higher vulnerability.

When an indicator presents different vulnerabilities in different areas of a watershed, the final indicator vulnerability score is calculated using the simple arithmetic average, with the percentage of each area being multiplied by the vulnerability score of the area (Formula 2).

$$Vu\ln erability\_Indicator_i = \sum_{i=1}^{n} Value_i * weight_i \qquad \text{(Formula 2)}$$

In Formula 2, “n” is the number of areas with different vulnerability values assumed by a particular indicator *i* in a watershed; “$Value_i$” is the normalized vulnerability value of indicator *i*; “$weight_i$” is the percentage of area presenting a particular vulnerability value for indicator *i* and “$Vulnerability\_Indicator_i$” is the final vulnerability value of indicator *i* in the watershed.

The simple arithmetic average is used to aggregate normalized vulnerability indicators in criteria, and the criteria in watershed vulnerability index. It is assumed that all indicators have the same importance in a particular criterion and that all criteria have the same importance in the formulation of the watershed vulnerability index.

iii) Phase environmental performance evaluation

The environmental performance evaluation of an innovation and of its substitute technology is performed in each life cycle phase. Initially, the values of the performance indicators gathered in the studied unit, usually related to a certain production mass, are adjusted to the production mass defined in the reference flow. A linear correlation is assumed between the production mass and the values obtained by the indicators in the field measurement.

In sequence, the indicators with potential to disturb the environment in a watershed scale are then multiplied by the EVI.

After adjusting and considering environmental vulnerability, the values of the environmental performance indicators are normalized to a standard non-dimensional scale. To normalize these indicators, the “maximum or minimum score” linear scale transformation, proposed by Malczewski (1999), is used. The “maximum score” transformation rule (Formula 3a) is used when “the higher the indicator value, the higher the environmental performance”, while the “minimum score” rule is used when “the higher the indicator value, the lower the performance” (Formula 3b). These rules allow the conversion of different indicators’ measurement units to a standardized score that ranges from 0 to 100, where 0 represents the worst performance and 100, the best.

$$Indicator\_normalized_i = \left(\frac{Indicator_i}{Value_{\max_i}} \right) * 100 \qquad \text{(Formula 3a)}$$

$$Indicator\_normalized_i = \left(\frac{Value_{\min_i}}{Indicator_i} \right) * 100 \qquad \text{(Formula 3b)}$$

In Formulas 3a and 3b, “$Indicator_i$” is the measured value of indicator *i* that was already adjusted and weighted by EVI and is related to an innovation or to its substitute technology; “$Value_{maxi}$” is the maximum value of indicator *i* and $Value_{mini}$” is the minimum value of indicator *i*, obtained by the comparison between the value assumed by the innovation and by its substitute technology; “$Indicator\_normalized_i$” is the normalized value of indicator *i*, when evaluating an innovation or its substitute technology.

The simple arithmetic average is used to aggregate normalized performance indicators in criteria, criteria in principles and in the phase environmental performance index. It is assumed that all indicators have the same importance in a particular criterion and that all criteria have the same importance in the formulation of principles and the final environmental performance index.

iv) Final environmental performance evaluation

Next, the values of each indicator, already adjusted and weighted by the vulnerability index, are aggregated into a total value that represents all life cycle phases. To aggregate the values assumed by an environmental performance indicator in each life cycle phase, one of two approaches are used: the sum of the values obtained by an indicator, when its measurement unit is related to mass, energy, volume and area; the simple arithmetic average of the values obtained by an indicator, for other measurement units (e.g. dS/m).

Finally , the same steps already described to a particular life cycle phase are followed, involving data normalization and aggregation, leading to the determination of the innovation and its substitute technology final environmental performance index.

Discussion and Conclusions

The presented Ambitec-Life Cycle method is a new approach to the environmental performance evaluation process of agro-industrial innovation. The method integrates life cycle thinking, vulnerability analysis, and the multi-criteria structure used by the Ambitec-Agro System, the current method being used for technology innovation impact assessments at Embrapa, Brazil.

From LCA theory, Ambitec-Life Cycle brought the expanded view that every product has a life cycle that must be considered when performing its environmental performance evaluation. The focus on just one phase of a product life cycle can mislead the performance evaluation of an innovation, because performance can be better in that single phase but worse in others. Hence, the environmental assessment of an innovation and its products, considering its entire life cycle, can reveal opportunities for technological improvements in all phases.

The proposed method also uses the LCA concepts of function, functional unit and reference flow that give a common base for comparison between an innovation and its substitute technology. This comparison is necessary because the intention is to promote the development and adoption of new processes and products that have a better environmental performance than existing ones. Without using these concepts, there is a risk of comparing technologies with little function resemblance and of gathering consumption and emissions data related to different quantities of the final products, making it difficult to interpret the results.

The vulnerability theory brought the perception that the magnitude of an impact depends on the ecological and socioeconomic characteristics of the area or ecosystem that provides the resources and receives the emissions related to a product life cycle phase. Analyzing the literature about the vulnerability concept, three main criteria were identified as important, at the watershed scale: exposure, sensitivity and capacity of response. The vulnerability analysis integrated in the Ambitec-Life Cycle method makes it feasible to simulate different scenarios for the innovation when adopted, according to the places where its life cycle may occur. This analysis can guide the innovation transfer process by revealing watersheds that are more or less vulnerable to a particular phase of an innovation life cycle.

From the Ambitec-Agro System, the proposed method brought the multi-criteria approach with the principles, criteria and indicators hierarchy. This favored the selection of criteria and indicators relevant to agro-industrial activities, their aggregation in sustainability principles and aggregation in a final environmental performance index.

In the environmental impact assessment study area, there is a large number of tools available that evaluate the environmental impact of development projects or policies, some that evaluate agro-industrial activities and a few that evaluate agro-industrial technological innovations. In this context, the Ambitec-Life Cycle method enriches the debate and the action in this area.

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Multicriteria comparison of RA and LCA toxicity methods with focus on pesticide application strategies

Kägi T.[1], Bockstaller C[2]., Gaillard G. [1], Hayer, F.[1], Mamy L.[3], Strassemeyer J.[4]
[1] Agroscope Reckenholz-Tänikon Research Station ART, CH-8046 Zurich, Switzerland,
[2] INRA UMR 1121 Nancy-Université – INRA, Agronomie et Environnement Nancy-Colmar, FR-68021 Colmar Cedex, France
[3] INRA UR 251 PESSAC Versailles, FR-78026 Versailles Cedex, France
[4] Julius Kühn-Institut (JKI) Bundesforschungsinstitut für Kulturpflanzen, DE-14532 Kleinmachnow

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Abstract

Several risk assessment (RA) and life cycle assessment (LCA) methods (EDIP97, EI99, IMPACT2002+, I-PHY, PRZM-USES, SYNOPS, and USES-LCA) to calculate the environmental impacts of pesticide use were evaluated. The evaluation scheme is mainly based on the work of the ITADA project COMETE (Bockstaller *et al*. 2006, 2007). It consists in a set of criteria divided into the three dimensions scientific soundness, practical feasibility and stakeholder utility, similarly to the OECD-Report on environmental indicators (OECD 1999). Criteria were assessed by the project group together with a cross-validation procedure. Considering the coverage of different relevant issues, the method PRZM-USES shows the best results for the coverage of environmental issues, human health and exposition pathways, followed by the LCA methods EI99, USES and Impact2002. Risk assessment methods SYNOPS and I-PHY yield lower results, because both do not consider human health. But the last mentioned is advantageous regarding coverage of agricultural branches and production factors and finally the method SYNOPS has strengths in geographical application, because very detailed data for field surroundings and climate are used. Regarding the aspects of practical feasibility and stakeholder utility the methods SYNOPS and I-PHY are advantageous compared to the other methods due to graphical user interface and implemented pesticide database, which reduces the time to fill in. Regarding the other methods, the differences are only minimal. As a next step of the assessment, all methods will be applied to case studies on pome fruit, maize and tomato to check if the expert opinions presented in this report are confirmed in practice.

Introduction

The life cycle inventory of a product often contains hundreds of substances, of which many have the potential to cause toxic effects on the environment, the ecosystem or human beings. Their life cycle impact is defined by characterisation factors for the ecotoxicity and human toxicity. Over the last years, many Life Cycle Assessment (LCA) models have been developed in order to analyse the toxic effect of chemical substances to environment and human health. As revealed by earlier comparative studies, these models vary substantially in their scope, applied modelling principles and in terms of the characterization factors they produce (Dreyer *et al*. 2003; Pant *et al*. 2005). Huibregts & Jolliet (2008a&b) compared and evaluated various models on midpoint level for aquatic and terrestrial ecotoxicity and human toxicity considering the environmental relevance and scientific robustness. Experience shows substantial variation between the models when looking at pesticides in agricultural production systems (Nemecek *et al*. 2005). To our knowledge there is no comparative study about the relevance of the toxicity methods when considering the pesticide use in agricultural systems.

The high number of pesticides available on the market and the modelling of the fate and effect of these pesticides make the handling of this question difficult. Current LCA methods can not consider all pesticides so far. Furthermore, the fate analysis in the methods is often rather simple in order to be able to assess chemicals with only few known properties. The recent announcement of the newly developed USEtox method (Rosenbaum *et al*. 2007) should improve the situation in LCA. But so far it is not known whether the improvements in USEtox will be sufficient enough for pesticide applications in agriculture.

While LCA characterizes emissions over a product's life cycle, it does not allow for a complete assessment of a product's potential impacts, also sometimes referred to as its "safety profile" or its risk assessment. This is because LCA reports emissions on a chosen functional unit basis (i.e. 1 kg finished product). Risk Assessment methods (RA) are designed to quantify the probability of adverse impacts for each type of emission, taking into account all sources of exposure. LCA was not designed to do that, but rather it was designed to understand the relative contribution of each stage of the life cycle to certain environmental impact categories. For these reasons, a closer collaboration between LCA and Risk Assessment (RA) modelling approaches has been done in order to profit from the RA developments.

Method / Approach

In the past, different criteria lists to compare agro-ecological methods were developed. Some authors use a descriptive (for example Girardin 2001; Reus *et al*. 2002; van der Werf and Petit 2002) others a more systematic approach (Gebauer and Bäuerle 2000; Hertwich *et al*. 1997). Following Bockstaller *et al*. (2007) these criteria lists do not include all aspects or are not transparent. Bockstaller *et al*. (2006) developed a new evaluation tool with clearly defined decision rules. The evaluation of risk assessment and LCA methods to calculate impacts of pesticide use presented in this report is mainly based on this work. Some adaptations have to be made to cover all aspects of the methods evaluated here. The evaluation is divided into three dimensions (scientific soundness, practical feasibility and stakeholder utility) similar to the one described in the OECD-Report on environmental indicators (OECD 1999).

The criteria list is derived from the work of Bockstaller *et al*. (2006) and Gaillard *et al*. (2005). The criteria are adapted to the evaluation of indicator methods assessing the impacts of pesticide in an LCA framework. Each author of the method or researcher supporting an indicator first filled in the tables of criteria. The method developers not represented in the European Network for the Durable Exploitation of Crop Protection Strategies (ENDURE) were separately consulted for supporting the evaluation. A cross-validation of the evaluation of each indicator has been done in order to avoid evaluation discrepancies.

Eleven criteria for the group scientific soundness are presented in Tab. 1. Five criteria refer to the coverage of the environmental issues (output), the production branches (domain of application) and the production factors (input). Three criteria tackle the construction of the indicators, the indicator type and the degree of process integration. The three last ones deal with the quality of the indicator in term of result (avoidance of incorrect conclusions) and implementation (transparency). Furthermore six criteria for the group practical feasibility and three for the group stakeholder utility are presented in Tab. 2. and Tab. 3 respectively.

Most of the sub-themes for the group’s practical feasibility and stakeholder utility are divided into three user groups (extension services, authorities and scientists); because it is assumed that their demands differ from each other.

The decision rules and more detailed values are given in Kägi *et al*. (2008). For all criteria the values can range from 1 to 5, whereby 1 stands for low and 5 for a good accordance between method and criteria.

The following seven toxicity methods were analysed according to above described procedure:

SYNOPS (Gutsche and Strassemeyer 2007)

I-PHY (Bockstaller *et al.* 2008)

PRZM-USES (Mamy *et al*., 2007)

EDIP97[2] (Hauschild and Wenzel 1998)

USES (Huibregts *et al*. 2000, Guinée *et al*. 2001)

IMPACT2002+[1] (Jolliet *et al*. 2003)

EI99[1] (Goedkoop and Spriensma 1999)

Results and Discussion

Scientific soundness

The group scientific soundness is divided into 11 categories with 1 to 13 subcategories. The category values in Tab. 1 represent the means of the respective subcategories.

Tab. 1: Criteria scores for the dimension scientific soundness (1 = low accordance, 5 = high accordance).

Criterion	**Score**							
	SYNOPS	I-PHY	EDIP	EI99	USES	Imp02	PRZM-USES	**Average**
Coverage of environmental issues	3.2	2.6	2.8	3.2	3.2	3.0	3.4	**3.1**
Coverage of human health	1.0	1.7	2.8	4.5	4.8	4.3	4.8	**3.2**
Coverage of exposition pathways	2.9	2.5	2.1	3.1	3.1	2.6	3.6	**2.8**
Coverage of agricultural production branches	3.8	3.7	2.0	2.0	2.0	2.0	3.7	**2.7**
Coverage of geographical application	4.2	2.0	1.0	1.2	1.2	1.2	1.5	**1.8**
Coverage of production factor	2.1	2.0	1.3	1.3	1.3	1.3	1.5	**1.6**
Indicator type, depth of environmental analysis	4.0	4.0	4.0	4.0	4.0	4.0	4.0	**4.0**
Integration of processes	4.0	4.0	3.0	4.0	4.0	4.0	4.0	**3.9**
Avoidance of incorrect conclusions linked to calculation method	4.0	4.0	3.0	4.0	4.0	4.0	4.0	**3.9**
Avoidance of incorrect conclusions linked to outputs	4.0	4.0	3.0	4.0	4.0	4.0	4.0	**3.9**
Transparency	4.0	4.0	4.0	4.0	4.0	4.0	4.0	**4.0**
Average	**3.4**	**3.1**	**2.6**	**3.2**	**3.2**	**3.1**	**3.5**	

Coverage of environmental issues

This category evaluates to which part the methods cover the entirety of environmental aspects. It is divided into 5 subcategories (aquatic risk (number of target species), aquatic risk (type of indicator), terrestrial risk (number of target species), terrestrial risk (type of indicator), risk assessment for beneficial organisms). All methods cover the aquatic risk satisfactorily. Looking at the terrestrial risk, the methods EDIP and I-PHY only partially cover this environmental issue because only one target

[2] Only the ecotoxicity and human toxicity methods of this methodology are considered.

species and only the chronic or acute risk potential is calculated for single products. On the other hand, I-PHY is the only method which considers the risk assessment for beneficial organisms.

Coverage of human health

This category evaluates to which extent the human health is regarded in the methods. It is divided into 4 subcategories (contamination of drinking water (pesticides uptake through water), risk for farmer during spraying (inhalation of pesticides), risk for harvester (pesticides on plants) and risk for consumers (pesticides uptake through food)). The low values for SYNOPS and I-PHY are due to the fact that SYNOPS is not designed for human health and I-PHY estimates effects on health only with one indicator (pollution of drinking water). The other methods cover pesticide uptake via food, drinking water (Impact2002) and in addition via inhalation (PRZM-USES, EI99 and USES).

Coverage of exposition pathways

This category considers how detailed the methods model the fate of the emitted substance, for instance the degradation and accumulation and the dispersion to the environmental compartments. The EDIP method shows the lowest performance for this category, because pesticide transfer to surface water by runoff, drainage, leaching and erosion, and pesticide degradation/accumulation are not considered. The most accurate estimation for the fate of substances is done by the PRZM-USES method, because this method regards nearly all aspects with exception of drift.

Coverage of agricultural production branches

This category assesses the possibility to apply the methods on different branches of the agricultural production regarding pesticide use. It is divided into 6 subcategories: arable farming, wine growing, fruit production, other special crops, pasture (plus fodder crops and permanent meadow) and animal production. Here two groups emerge. Less than 50 % of the used pesticides are characterized for the LCA methods EDIP, EI99, USES and Impact2002+, whereas for the RA methods SYNOPS, I-PHY and PRZM-USES in most cases 75-95 % of the pesticides used in the different branches are characterized.

Coverage of geographical application

This category concerns the geographical variance of field parameters and how it is dealt within the models. It is divided into 3 categories (field specific parameters, parameters for field surrounding and climate data) with 3 to 6 subcategories. The SYNOPS method has the best performance. Compared to the other evaluated models SYNOPS considers environmental field parameters and the surrounding of the field using GIS functionalities by linking the model to geo-referenced databases for land use, soil conditions and climate data and to a dataset of regionalized surveys of pesticide application. The other methods mostly use only field specific parameters and to some extent also climatic (PRZM-USES) or field surrounding (I-PHY) parameters.

Coverage of production factors

In this category the methods are analyzed regarding the implementation of pesticide storage, handling and application into the models. The coverage of production factors is divided into two categories (farm level and field level) with 3 and 8 subcategories respectively. The methods EDIP, EI99, USES, Impact2002 and PRZM-USES only take the application rate of active ingredients and partly the formulation of the product into account, whereas SYNOPS and I-PHY also regard the type of sprayer, the sprayed area (partial application), mitigation measures (injection, incorporation of pesticides, etc.), the implementation of fixed buffer strips and the implementation of product dependent buffer zones/strips at the field level. At farm level none of the methods considers possible impacts due to the storage of pesticides, infrastructure for filling and washing the sprayer and waste management (packaging, rest of pesticides).

Indicator type, depth of environmental analysis

This category assesses the indicator type used according to the DPSIR (Driving forces, Pressures, States, Impacts and Responses)framework. All methods use indicators for assessing potential impact based on emissions.

Integration of processes

This category assesses the integration of the detailed processes for fate and exposure into the models. The method EDIP uses empirical approaches to calculate the fate and exposure, whereas the other methods use more reliable conceptual or mechanistic models.

Avoidance of incorrect conclusions linked to calculation method and outputs

All methods are based on validated models. For the majority of the models the risk is low to draw wrong conclusions either due to the model calculation or due to a wrong interpretation of the output. The only exception is EDIP which is based on expert recommendations rather than on models.

Transparency

All methods are transparent and detailed information on calculations or reference values are available.

Criterion "practical feasibility"

The criterion practical feasibility is done for three user groups. As the three user groups extension services, authorities and scientists show similar tendencies, only the group scientists is presented in Tab. 2. In general the methods are most suitable for scientists followed by authorities and extension services. In practice it is very difficult to estimate the practical feasibility for the single groups. The calculation tool used in the methodology EI99 was not available and therefore all values are set to 1.

Accessibility of input data

This category assesses the availability of data for different data groups (Meteorological data, overview of field characteristics, pesticide properties and field specific data). For the methods SYNOPS, I-PHY and PRZM-USES, the data are easier to access as for the methods EDIP, USES and Impact2002, because they have implemented databases. The data accessibility is lower for the user group extension services than for authorities or scientists, because this group does not have the same access to pesticide properties than the other ones.

Qualification requirements

For extension services the main problem is the qualification requirement. For all methods an advanced training is needed for data collection, calculation or programming the input files and interpretation. The PRZM-USES method has the highest requirements (more than one week is needed to learn how to use the models). The methods SYNOPS and I-PHY have the lowest requirements, because they are software-based with predefined input options.

External service

This category considers the necessity of an external service if using the method. The assessment strongly varies according to the target group designated and the assumption that we have to take about the technical and scientific self-sufficiency. All methods show the same trend. The lowest rates are achieved for the target group "authorities" and the highest for the one "scientists".

User-friendliness

The methods SYNOPS and I-PHY are the most user-friendly, because they use a graphical user interface with predetermined input options and illustrated results. All other methods are lacking these options.

Support

The support of SYNOPS is suboptimal to the one offered by the other methods, because only an example is available, whereas for all other methods a guideline is also present.

Time needed to calculate/fill in

For SYNOPS and I-PHY the least time is needed to fill in and calculate as a database for the active ingredients is implemented in the software. The longest time is needed for the PRZM-USES, because the models of the method have to be parameterized. The time needed to calculate the other methods is in between for no parameterization has to be done, but also no database is implemented.

Tab. 2: Criterion “practical feasibility”: list of themes to score on a scale between 1 and 5 (1 = low accordance, 5 = high accordance).

Practical feasibility **User Group (scientists)**	**score (1 to 5)**							
	SYNOPS	I-PHY	EDIP	EI99	USES	Imp02	PRZM-USES	**Average**
Accessibility of input data	5	5	4.5	1	4	4	4.7	**4.0**
Qualification requirements (user)	3	4	2.3	1	2.3	3	3	**2.7**
External services	3	3	5	1	5	3	5	**3.6**
User-friendliness	3	3	1	1	1	1	1	**1.6**
Support	3	4	4	1	4	4	3	**3.3**
Time needed (to calculate/ fill in)	5	5	3	1	3	3	3	**3.3**
Average	**3.7**	**4**	**3.3**	**1**	**3.2**	**3.0**	**3.3**	

Criterion “stakeholder utility”

Likewise for practical feasibility, the criterion stakeholder utility is divided into three user groups of which only the group scientists is presented in Tab. 3 due to similar tendencies. All methods meet the needs of all three user groups to a high degree, because all could be applied to different spatial areas and could be used to compare strategies policies and scenarios at different levels (farm, regional). The methods SYNOPS and I-PHY are more advantageous in terms of unambiguousness and communicability of results since the results are presented with more details (for example graphical illustrations and reference values) than in EDIP, USES, Impact2002 and PRZM-USES where only a scientific value is given..

Tab. 3: Criterion “stakeholder utility”: list of themes to score on a scale between 1 and 5 (1 = low accordance, 5 = high accordance).

Stakeholder utility **User group (scientist)**	**score (1 to 5)**							
	SYNOPS	I-PHY	EDIP	EI99	USES	Imp02	PRZM-USES	Average
Coverage of needs	4	5	4	4	4	4	4	**4.1**
Unambiguousness of results	3	3	1	1	1	1	1	**1.6**
Communicability of results	3	4	1	1	1	1	1	**1.7**
Average	**3.3**	**4.0**	**2.0**	**2.0**	**2.0**	**2.0**	**2.0**	

Discussion and Conclusion

The seven methods EDIP97, EI99, IMPACT2002+, I-PHY, PRZM-USES, SYNOPS and USES-LCA were assessed for pesticide consideration (eco- and human toxicity) in Risk Assessment (RA) and Life Cycle Assessment (LCA).

Considering the environmental issues, all methods cover the aquatic risk satisfactorily. Looking at the terrestrial risk, SYNOPS covers this environmental compartment best considering the acute and chronic risk potential of two target species. The methods EDIP and I-PHY only partially cover this environmental issue because only one target species and only the chronic or acute risk potential is calculated for single products. The terrestrial ecotoxicity in IMPACT2002+ is based on aquatic data and therefore it should be seen as a rough estimate. On the other hand, I-PHY is the only method which considers the risk assessment for beneficial organisms, an important issue in integrated pest management.

Considering human toxicity, the methods SYNOPS, I-PHY and EDIP do not cover this aspect sufficiently. SYNOPS does not consider human toxicity at all; I-PHY does not consider the risk for farmers and consumers, whereas EDIP does not consider the pesticide uptake through groundwater. Huijbregts & Jolliet (2008b) report a similar result, namely a lower score for EDIP compared to other ecotoxicity methods. On the contrary, the methods USES, IMPACT2002+, EI99 and USES-PRZM face this aspect almost entirely.

In view of the exposition pathways, EDIP only roughly estimates the fate factors to water, air and soil and therefore shows the lowest scores. A similar result is reported by Huijbregts & Jolliet (2008a) for EDIP compared to other ecotoxicity methods. The methods SYNOPS, EI99, USES and USES-PRZM reach the highest scores since calculating the exposition pathways is well founded.

There is an apparent advantage of the RA methods over the LCA methods for the criteria sets coverage of agricultural applicability and coverage of production factor. The LCA methods can so far only handle a limited number of pesticides. Furthermore, they are not detailed enough to consider production management aspects or processes on the field such as incorporation etc., whereas the RA methods are especially designed for assessing pesticide applications.

Considering the coverage of geographical applicability, the method SYNOPS shows a clear advantage over the other methods which do not cover this aspect satisfactorily.

Looking at the other criteria sets such as the depth of analysis, the integration of processes, the avoidance of incorrect conclusions and transparency, there is no difference between the methods. All methods cover these aspects adequately.

Regarding the aspects of practical feasibility and stakeholder utility the methods SYNOPS and I-PHY are advantageous compared to the other methods. First, they are available as a software tool with a graphical interface, which facilitates the handling. A second point is that a database of active ingredients and pesticide products is implemented in the software of both methods. Therefore they are much more user-friendly and the input data is easier to access than for other methods. Finally for both methods the outputs are presented in figures with some reference values, which simplify the interpretation and communicability of the results.

As a result of this assessment, it appeared that each method has its strengths and weaknesses Nevertheless it also emerged that the RA methods generally have a better scientific soundness and stakeholder utility than the LCA methods. This can be explained to some extend by their design of quantifying the probability of adverse impacts taking into account all sources of exposure. The best methods are SYNOPS and PRZM-USES, whereas EDIP has generally the lowest values. EI99 and USES are the LCA methods with the highest scientific soundness, whereas EDIP and USES are the most practical feasible methods. As a general result of the described comparison it can be stated that the method SYNOPS is most suitable for the toxicity estimation of pesticide strategies (with the exception of human toxicity which is not yet designed). Concerning only LCA methods the method USES performs best. However the statements of SYNOPS and the other RA methods do not completely comply with the philosophy of the LCA. More research is needed to interlace the RA methods into the LCA methods.

As a next step of the assessment, all methods will be applied to three case studies (pomefruit, maize and tomato) according to the methods outlined in Bockstaller *et al*. (2006, 2007) and Gaillard *et al*. (2005) to check if the expert opinions presented in this report could be confirmed and to compare similarities in the results. The presented approaches and further concepts to combine different tools in LCA toxicity methods need to be followed up.

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Comparative Assessment of the Potential Impact of Pesticides Used in the Catchment of Lake Geneva

PJ. Copin[1], N. Chèvre[1], R. Charles[3], A. Klein[4], M. Margni[2]
[1]IPTEH, Amphipôle, Faculté de Géosciences et de l'environnement, Université de Lausanne, CH-1015 Lausanne, Switzerland, Pierre-Jean.Copin@unil.ch.
[2]CIRAIG, Département de génie chimique, École Polytechnique de Montréal, C.P. 6079, Montréal (Qc) Canada, H3C 3A7.
[3] Station de recherche, Agroscope-Changins-Wädenswil ACW, Case postale 1012, 1260 Nyon 1, Switzerland.
[4]CIPEL, Route de Duillier, Case postale 1080, CH – 1260 Nyon 1, Switzerland.

Keywords: Life Cycle Impact Assessment, Comparative Risk Assessment, Ecotoxicological Impact, Pesticides, Multimedia Modeling, Model Validation.

Abstract

Whilst pesticides historically appeared beneficial, the harmful side effects of some of them were rapidly highlighted. The analyses of Lake Geneva's water (Switzerland) during the last years revealed the presence of approximately 30 pesticides, mainly herbicides and fungicides, at almost all water depths. In this study, we demonstrated the feasibility of applying a Comparative Risk Assessment modelling framework, to provide additional understanding in the complex issue of pesticide contamination on locally affected areas. The applied methodology combines a fate and an ecotoxicological effect assessment to evaluate the potential risk of a pesticide emission. The fate factor links a substance emission, e.g. into air, to an increase of the concentration in an environmental media, e.g. fresh water. Model predictions were computed using the IMPACT 2002 model and compared with monitored results and differences where discussed. The effect factor links this concentration increase to a loss of living species. Two methods to calculate the aquatic effect factor were compared: the Assessment of the Mean Impact (AMI) method based on the mean ecotoxicological response of available species and the Species Sensitivity Distribution (SSD) method based on the most sensitive species of the ecosystem. The impact score, obtained combining the fate and the effect factors, were used to compare the potential impacts of pesticides applied to the catchment of Lake Geneva.

Our study showed that the obtained risk based indicator can be used to support decision-making, helping i) local authorities to identify the key pesticides of concern and identify and characterize additional sources of pollution and ii) farmers to promote good agricultural practices.

Introduction

The lemanic watershed has an area of 7 975 km^2 and is located in the western part of Europe, in Switzerland and France. One of the main characteristics of the watershed is the presence of Lake Geneva. Its surface is 580.1 km^2 for a volume of 89 km^3 and a mean depth of 152.7 m. It was formed by the retreat of the Rhone glacier, 15 000 years ago. Lake Geneva receives water from many rivers originating from different regions of Switzerland and France and is crossed from east to west by the Rhone River. This latter is the most important contributor in terms of volume of water. Around 1 million people live in the lemanic region and about 50% of this population is connected to the network supplied by Lake Geneva's water. The large towns surrounding the lake are Geneva, Lausanne, Thonon-les-Bains and Evian (Fig. 1).

For the Swiss part of the lemanic watershed, a large fraction of soils is allocated to agriculture (20.5 % of cultivated fields and 23.0 % of pasture), another fraction to forest (22.0%) while 34.5 % of lands are uncultivated. The main cultivations are grass species (63.1 %), open lands (26.1 %) and wine (6.6 %). In addition, intensive orchards (2.6 %) and market gardener crops (1.0%) are present.

During the last decades, Lake Geneva has been subjected to various forms of pollution. For a long time, phosphate concentrations were far above the limits defined by Swiss law. The situation has improved since the eighties, when phosphate was banned in Swiss laundry soaps and restricted in French washing powders. Efforts to curb this type of pollution at its sources must be carried on and intensified, especially in sectors where phosphate concentrations remain above the regulatory limits (CIPEL 2007). However, organic micropolluants continue to be a serious problem for Lake Geneva, particularly in the case of pesticides and pharmaceuticals. The latest measurements reveal an increase in the number of detected pesticides in water. They arise from agricultural, gardening, industrial and urban activities. In the whole lemanic catchment, 182 pesticides are used in agriculture where the most contributing cultivations are arboriculture and viticulture (CIPEL 2007). These pesticides may reach surface water through rainwater runoff or groundwater. Their occurrence in Lake Geneva may impact the lake’s ecosystem. It is therefore crucial to develop and validate tools which evaluate the potential risk of pesticide emission in this catchment to support local authorities in setting emission control & reduction priorities and farmers to promote good agricultural practices. In this paper, we apply a comparative risk assessment framework aiming at identifying the most problematic pesticides emitted within the Geneva lake’s catchment. This approach has been evaluated against measurements.

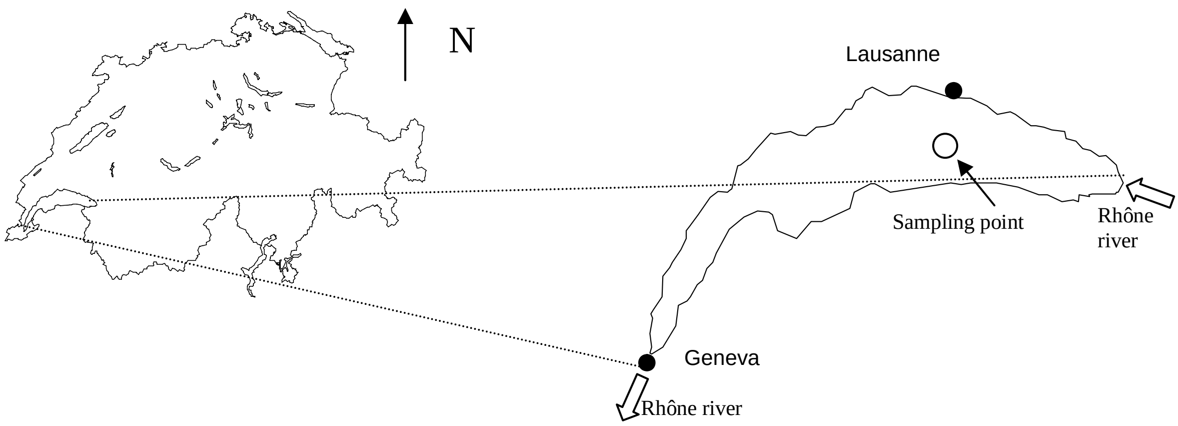


Fig. 1: Lake Geneva and sampling point

Method / Approach

Comparative risk assessment in LCA

Risk assessment in comparative frameworks, such as Life Cycle Impact Assessment (LCIA) is performed by comparing impact scores on an aquatic ecosystem of several substances, like pesticides. This impact score can be represented by the combination of a so-called substance specific characterization factor and its emitted amount (Hertwich *et al.*, 2002). The characterization factor is the combination of the fate factor and the effect factor:

$$S = CFa_i * M_i = FF_i * EF_i * M_i \text{. (Eq. 1)}$$

With S: Impact score of the substance [PAF.m^3.year] (PAF is the Potentially Affected Fraction of species due to exposure to the chemical),

CFa_i: Characterization factor of the substance *i* for the aquatic ecosystem [PAF.m^3.year/kg],

FF_i: Fate factor calculated for a unitary emission of substance *i* [year],

EF_i: Effect factor of the substance *i* [PAF.m^3/kg],

M_i: Quantity of emitted substance [kg].

The adopted modelling emission approach to ecosystem damage is described in Fig. 2 (adapted from Pennington *et al*. 2006). The fate factor links the quantity of a substance released into the environment to the chemical masses (or concentrations) in a given compartment. A multimedia fate model based on steady-state mass balance equations, including degradation intermedia transfer rates between environmental compartments (air, water, soil, sediment, etc.) is used for this purpose. The effect factor relates the concentration increase in a compartment, namely water, to the damage of the concerned ecosystems in term of potentially affected fraction of species (Payet 2004).

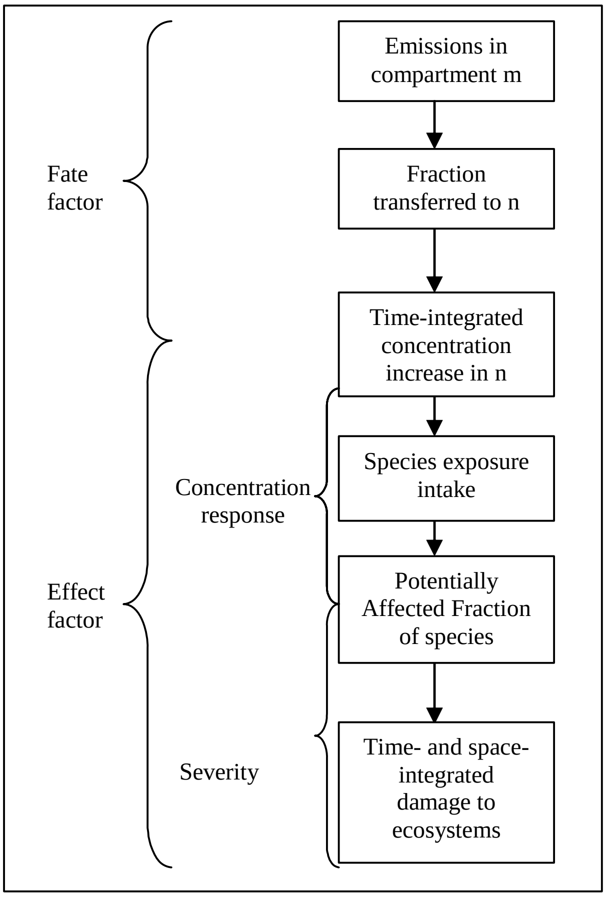


Fig. 2: Diagram of impact evaluation in LCA for ecotoxicity (adapted from Pennington *et al*. 2006).

IMPACT 2002

The source to damage framework is modelled with impact IMPACT 2002, a multimedia fate and multipathway exposure model for Western Europe (Pennington *et al*. 2005). This model allows estimating the concentrations of chemicals in different compartments (air, water, soil, sediment, vegetation) and calculating the impact on human health and ecosystems. IMPACT 2002 was adapted to the lemanic watershed by taking into account the landscape parameters of this region: area of the lemanic catchment, water depth, area of Lake Geneva, etc.

IMPACT 2002 exists in two versions: a non-spatial and a spatially resolved model. In the spatially resolved model, the transport of chemical takes place between different compartments in the same media or between compartments belonging to different media. In the non-spatial model, also called box model, each media is represented by a single compartment. The transfer rate coefficients k (1/time) characterizes the different transport modes and loss rates. In this study, the non-spatial version of the model was applied. Results for each chemical are provided by a matrix format, which enables a very effective interpretation as suggested by Rosenbaum and colleagues (2007).

The authors describe how the fate factor F_{ij} - linking an emission from an initial compartment i to a receiving compartment j (air, water, soil, sediment and vegetation)- can also be formulated as a transferred fraction, f_{ij} [-], from an emission compartment i to a receiving compartment j, multiplied by the effective residence time in the final compartment j, θ_{jj} [h]. The residence time corresponds to the negative inverse of the transfer rate coefficient matrix and the transferred fraction f_{ij} can be calculated as the ration between off-diagonal and diagonal elements of the fate matrix (see Rosenbaum *et al*. 2007 for more details).

$$FF_{ij} = f_{ij} * \theta_{jj} \Rightarrow f_{ij} = \frac{FF_{ij}}{FF_{jj}} \qquad \text{(Eq. 2)}$$

Fate factor for an agricultural application

In order to evaluate the prediction of the model for the lemanic region against monitored data, a fate factor for an agricultural application was calculated by weighting the fate factors FF_{wa} and FF_{ws} of the fate matrix by the respective emission on cultivations. Indeed, for an agricultural application, 15% of a pesticide is emitted directly in the air and 85% on soil surface (Humbert *et al*. 2007). Therefore, the fate factor from its emission in agriculture to its final compartment (water) is calculated by the following relationship:

$$FF_{weighted} = 0.15 * FF_{wa} + 0.85 * FF_{ws} \qquad \text{(Eq. 3)}$$

The fate factor calculated by the model (Eq. 3) may be converted into concentration (Eq. 4).

$$Concentration[kg/m^3] = \frac{Quantity\ of\ pesticide\ emitted\ [kg/h]}{Lake\ volume[m^3]} * FF[h] \qquad \text{(Eq. 4)}$$

These modelled concentrations were then compared with monitored concentrations measured at the sampling point in Lake Geneva (Fig. 1). These concentrations have been measured at different depths of Lake Geneva and at different times of the year during 2004 and 2005 (Edder *et al*. 2007).

Specific physicochemical parameters of the pesticides studied were collected as well including: K_{ow} coefficient, molecular mass, Henry's constant, tropospheric degradation half-life, water-column degradation half-life, sediment degradation half-life, vegetation and soil degradation half-life. Consequently, an assessment of the model was conducted for different types of pesticides, comparing the concentrations obtained with IMPACT 2002 to the concentrations measured at the sampling point of Lake Geneva (Fig. 1). The physicochemical parameters for each pesticide were found in the literature and the quantity of pesticide applied in the whole catchment was provided by the International Commission for the Protection of Lake Geneva (CIPEL, www.cipel.org).

Effect factor: AMI and SSD methods

The AMI method (Assessment of the Mean Impact) is a recently developed technique (Payet 2004) to derive the effect factor used in comparative risk assessment in LCA. The first step consists in obtaining the ecotoxicological data for a specific chemical on several species:

- EC50: Effect Concentration 50%; usually determined by laboratory testing for acute toxicity. It expresses the pollutant concentration at which 50% of the exposed organisms show the tested effect. Mortality is generally used as the indicator. EC50 data used in this project were collected in AQUIRE[3]
- NOEC: No-Observed Effect Concentration; concentration usually determined by laboratory testing for chronic toxicity. It expresses the pollutant concentration at which no effect is observable. Usually reproduction or growth is used as the indicator.
- LOEC: Lowest Observed Effect Concentration; the lowest concentration at which an effect is observable.

Thereafter, the data was converted as follows: Log_{10} transformation, extrapolation of EC50s from NOECs and LOECs, calculation of the EC50 geometric mean for every species, separating acute and

[3] http://cfpub.epa.gov/ecotox/

chronic data. If there are more than 3 chronic data covering at least 3 taxons, the chronic data must be used. If not, 3 acute data covering 3 taxons will be used. If none of these values are available, either the QSAR model (Quantitative Structure Active Relationships) is applied or the ecotoxicological dataset have to be improved by testing. Finally, the geometric mean for chronic and acute values ($HC50_{EC50\ ch}$ or $HC50_{EC50\ ac}$) is calculated and used to generate the effect factor. The effect factor, EF, is the change in the Potentially Affected Fraction of species induced by an increase in contaminant concentration [PAF.m^3/kg].

$$EF_c = \frac{0.5}{HC50_{EC50\,ch}} \ or \ EF_a = \frac{0.5}{HC50_{EC50\,ac}} \qquad \text{(Eq. 5 and 6)}$$

An alternative method was tested to assess the EF. It is derived from Species Sensitivity Distribution (SSD), in which curves represent the percentage of species affected as a function of the pollutant concentration (log NOEC or log EC50; Posthuma *et al*. 2002). For pesticides, the ecotoxicological data for the taxon that is the most sensitive to the treated chemical must be used to determine the SSD. Indeed for these substances, the SSD curve obtained was no longer unimodal but bimodal when all species were taken into account (Scheringer *et al*. 2001). Note that it is necessary to have a minimum of 10 ecotoxicological data in order to obtain a meaningful SSD curve (Solomon *et al*. 1996).

As not enough data is usually available to construct SSD-chronic, Chèvre *et al*. (2006) recently proposed a method to derive SSD-chronic from SSD-acute. This method was used in this study.

The Hazardous Concentration 30% (HC30), which is the concentration affecting 30% of the most sensitive organisms, was extrapolated. The level of 30% was chosen because we assumed that all ecosystems in agricultural regions have 30% of their species affected by other stressors (Posthuma *et al*. 2002). The sensitive species were considered to belong to this fraction. Therefore, the effect factor was calculated in the following way:

$$EF = \frac{1}{HC30} \qquad \text{(Eq. 7)}$$

Results

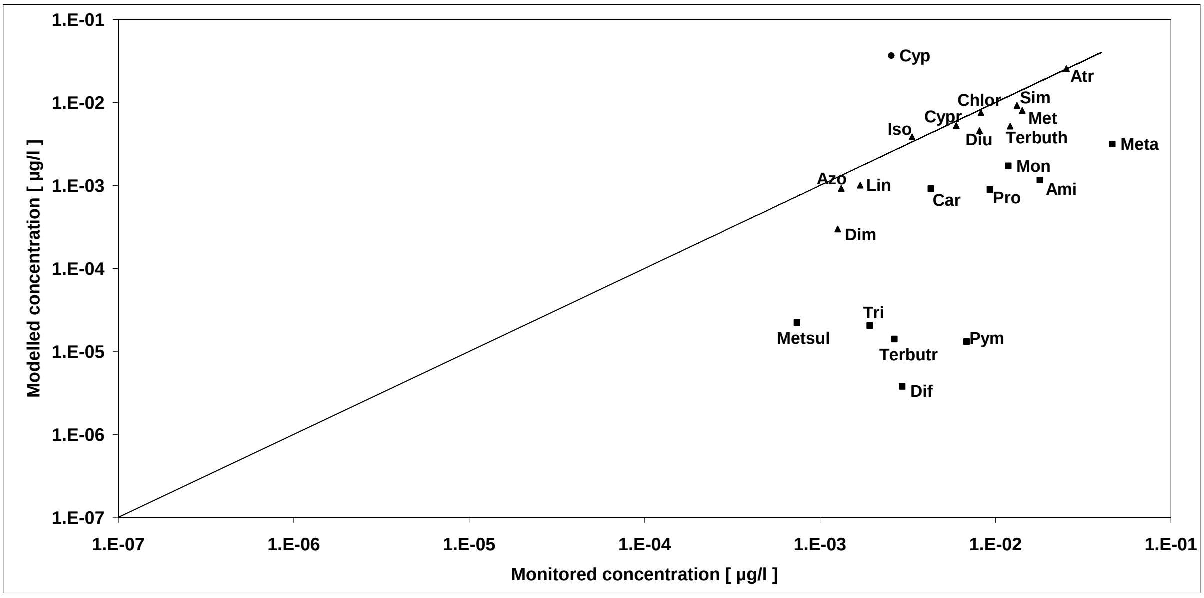
Assessment of the IMPACT 2002 model

The results of the model assessment are shown below (Fig. 3).

The straight line in Fig. 3 corresponds to the situation where modelled concentrations are equal to monitored concentrations. There are 11 pesticides for which modelled concentrations are similar to monitored concentrations (illustrated by triangles in Fig. 3). These pesticides are Dimethenamid, Linuron, Isoproturon, Chlortoluron, Simazine, Metolachlor, Diuron, Terbuthylazin, Atrazine, Azoxystrobin and Cyproconazol.

The modelled concentration of only one substance (Cyprodinil) is higher than its monitored concentration, while the values for 10 substances are smaller than their monitored counterparts (illustrated by squares in Fig. 3). They include: Carbendazim, Metalaxyl, Difenoconazol, Pymetrozin, Propiconazol, Metsulfuron-methyl, Triasulfuron, Amidosulfuron Terbutryn and Monolinuron. A large number of these pesticides are either fungicides or a type of herbicide (sulfonylurea).

The difference between modelled and monitored concentrations is discussed below.



Cyp	Cyprodinil	X	Chlor	Chlortoluron	V	Car	Carbendazim	X	Dif	Difenoconazol	X
Dim	Dimethenamid	V	Sim	Simazine	V	Mon	Monolinuron	X	Pym	Pymetrozine	X
Azo	Azoxystrobin	V	Met	Metolachlor	V	Ami	Amidosulfuron	X	Pro	Propiconazole	X
Lin	Linuron	V	Diu	Diuron	V	Meta	Metalaxyl	X	Terbutr	Terbutryn	X
Iso	Isoproturon	V	Terbuth	Terbuthylazin	V	Metsu	Metsulfuron-methyl	X	-	-	
Cypr	Cyproconazole	V	Atr	Atrazine	V	Tri	Triasulfuron	X	-	-	

Fig. 3: Correlation between modelled and monitored concentrations of several pesticides applied in the catchment of Lake Geneva. Abbreviations of the substances are explained in the table below; V corresponds to substances correctly assessed, X to poor evaluation.

Comparison of effect factors calculated with the AMI and the SSD methods:

To illustrate the results of the effect factors obtained with both methods, 2 herbicides were used, Atrazine and Simazine. These substances were selected because an important number of ecotoxicological data are available in the literature, particularly for sensitive species. The number of values regarding sensitive species for Atrazine and Simazine is respectively 146 and 24. The amount of values is higher than the standard of 10 prescribed for the application of the SSD method (Solomon *et al*. 1996). For the AMI method, there are over 3 chronic data covering at least 3 taxons.

Finally, using the 2 methods, the following effect factors were derived:

Tab. 1: Effect factors for Atrazine and Simazine using SSD and AMI methods.

	SSD Method		AMI method	
	HC30 [µg/l]	EF [PAF.m^3/kg]	HC50 [µg/l]	EF [PAF. m^3/kg]
Atrazine	7.6	131	524	954
Simazine	21.4	47	1280	391

Results for both methods lead to the same conclusions: the effect factor for Atrazine is higher than that of Simazine. With the SSD method, Atrazine is 2.7 times more toxic than Simazine in aquatic ecosystems, whereas with the AMI method Atrazine is 2.4 times more toxic than Simazine. Therefore, for Atrazine and Simazine, both methods can be used, as the ratio between the 2 effect factors is very similar.

Discussion

Fate factor and IMPACT 2002

The results show that the concentrations modelled with IMPACT 2002 are correctly assessed for 50% of the substances. For the majority of the other pesticides, the modelled concentrations are underestimated. For one substance (Cyprodinil), the modelled concentration is overestimated. Several reasons may explain these results:

(i) First of all, 3 substances tested in the model are also used in non-agricultural application as biocides (Kupper *et al*. 2005). Biocidal products can be contained in disinfectants, preservatives, agents for pest control or antifouling products (Lassen *et al*. 2001). These substances are mostly applied in cities and they are frequently found in urban water systems. Terbutryn, Propiconazole and Carbendazim, which were used to assess the model, are actually applied as biocides in cities around Lake Geneva. This would explain why the modelled concentrations (based only on agricultural contribution) are lower than the monitored concentrations. However, a better estimation is not possible because the amounts of used biocides are unknown. Consequently, this analysis of the results allows concluding that the presence of these 3 pesticides is certainly due more to their use as biocides than for agricultural application.

(ii) Secondly, five substances, whose modelled concentrations are smaller than the monitored concentrations, are manufactured in some industries located in the lemanic watershed. These substances are Amidosulfuron, Metsulfuron-methyl, Triasulfuron (Sulfonylurea), Pymetrozine and Metalaxyl (Bernard *et al*. 2007). Similarly to the biocides from urban area, the amounts of pesticides produced from the industries are unknown. This may be a reason why the model underestimates the concentrations. Thus, it can also be concluded that the presence of these 5 substances in Lake Geneva is mainly due to their manufacture in the concerned industries than to their application in agriculture.

(iii) Finally, Monolinuron, Difenoconazol and Cyprodinil are used only in agriculture. The difference between their monitored and modelled concentrations may be explained by the poor estimation of the half-lives in water. For example, in different databases, Monolinuron and Difenoconazol are defined as stable substances in water. Generally, this term of stability entails that half-life in water is higher than 30 days, meaning that the fate factor and the resulting modelled concentrations are underestimated by the use of underestimated half-lives. However, for Cyprodinil, the modelled concentration is overestimated. This could be due to an overestimation of its half-life in water. For the 3 substances, new laboratory testing of the half-lives is needed to verify or to obtain these values.

A detailed analysis of the rate coefficient matrix $\bar{k}$ allows determining the main loss process of a substance, advection, degradation or intermedia transfer to other media. The loss fraction of each process in each media is given by dividing each element of a column of the rate coefficient matrix by the diagonal element of this same column (Rosenbaum *et al*. 2007). To illustrate this kind of analysis, 2 substances are considered: Atrazine and Linuron. The results describing the main loss rates are presented in the tables 3 and 4 for 5 media: air, water, soil, sediment and vegetation.

Tab. 2: Losses by transport or advection-degradation for Atrazine

Atrazine	Air	Soil	Water	Sediment	Vegetation
Intermedia transport to other media :	24%	69%	0.3%	98.6%	1%
Advection and degradation in media :	76%	31%	99.7%	1.4%	99%
Total	100%	100%	100%	100%	100%

Generally, Atrazine is applied on the soil and then moves toward the water compartment. As expected, Atrazine present in soil is mainly removed by transport (69%). For Atrazine, the degradation and the advection losses are the dominating process particularly in water (99.7%) but also in air and vegetation.

Tab. 3: Losses by transport or advection-degradation for Linuron

Linuron	Air	Soil	Water	Sediment	Vegetation
Transport since media :	18%	24%	0.2%	21%	12%
Advection and degradation in media :	82%	76%	99.8%	79%	88%
Total	100%	100%	100%	100%	100%

For Linuron, the degradation and the advection losses are the dominating removal processes in all media, although the transported fraction from air, soil and sediment is significant.

Comparing the 2 substances, one can explain why the concentration in Lake Geneva is lower in Linuron than in Atrazine for a unitary emission; indeed, Linuron has a lower probability than Atrazine of reaching the media water.

Effect factor: comparison of the AMI and the SSD methods

The main difference between the two methods is the sensitivity of the species taken into account. The AMI method considers that all species of the ecosystem are affected by several stressors. Consequently, the pesticides will have an impact on all species and not only on the most sensitive ones.

Contrarily, for the SSD method, the focus is only on the most sensitive species. The hypothesis underlying the method is that when sensitive species are affected, there is a risk that the whole ecosystem is affected. As expected, SSD curves are better approximated using ecotoxicological data on sensitive species than with all species of the studied ecosystem. In Fig. 4A, when only sensitive species are used, the value distribution is a unimodal curve. On the other hand, in Fig. 4B illustrating all species, the data distribution is a bimodal function. However, the inferred curves are unimodal. Therefore, plots based on sensitive species are preferred since they provide better approximation curves.

A disadvantage of the SSD method is the assumption according to which 30% of species of ecosystems are affected by stressors (Posthuma *et al*. 2002). It must be pointed out that the structure of the lemanic ecosystem is not exactly known, hence, the percentage of affected species for Lake Geneva catchment cannot be considered as equal to 30% without doubt. The choice of taking HC30 to determine the effect factor is thus questionable and uncertainty remains regarding this value. Fieldwork would be necessary to determine this percentage of affected species for the lemanic watershed.

For the determination of the effect factor, we used two herbicides whose ecotoxicological data are readily available. Further testing using pesticides with less ecotoxicological data is therefore required to verify that the SSD method is valid.

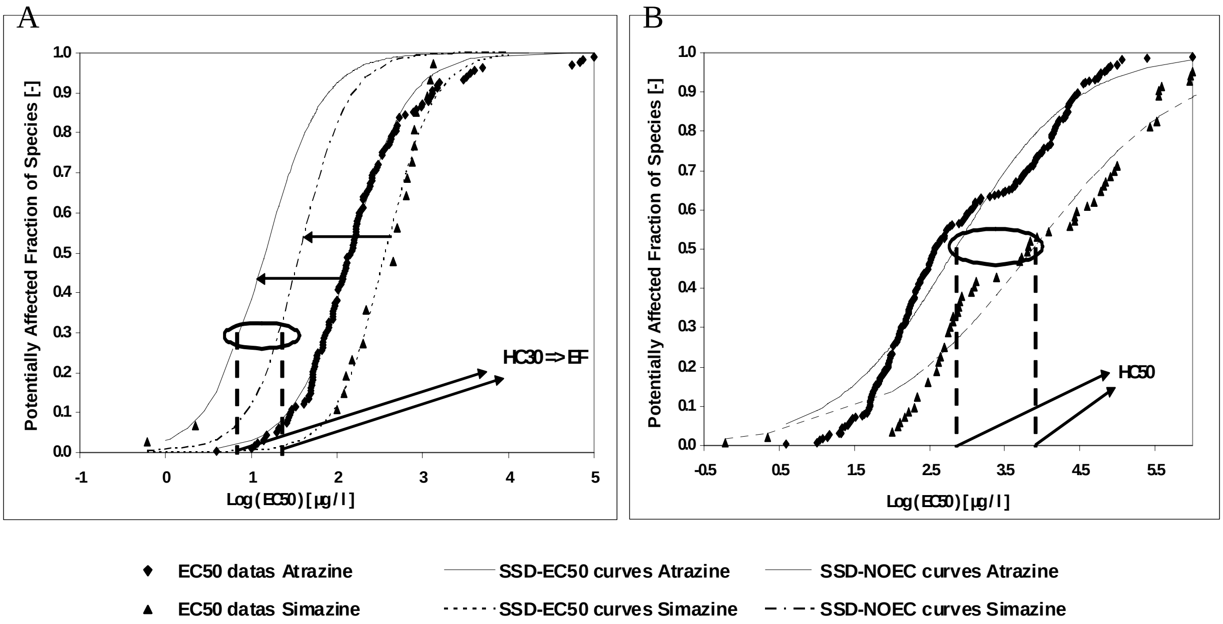


Fig. 4: Illustration of the SSD method used to calculate the effect factor required to evaluate LCIA on ecosystems for Atrazine and Simazine. The first graph (A) presents the method based on the most sensitive species and HC30 (Concentration causing a hazard for 30% of sensitive species). The second graph (B) presents the same method taking into account all species and HC50 (Concentration causing a hazard for 50% of all species).

Results of the source-to-impact characterization

The characterization factors of Atrazine and Simazine are presented below in Tab. 4. The CFa of Atrazine, which is the product between the fate factor and the effect factor (see Eq.1), is about 4 times higher than the one of Simazine, due to a higher toxicity (factor 3 on the EF). The fate factor of Atrazine, i.e. the capacity of a chemical to be transferred to water, is about twice the one of Simazine. The fate factor is a combination between the fraction of a chemical emission that is transferred to water multiplied by the residence time in water θ_{ww} (see Eq.2). As the transferred fraction to water is similar for both chemicals, the difference is only due to the residence time, which is ultimately driven by the degradation in water, being the residence time of the Geneva Lake up to $1.16{\cdot}10^5$ hours.

Tab. 4 : Fate factor, Effect factor and Characterization factor for Atrazine and Simazine

Substances	θ_{ww} [h]	FF_{ws} [h]	FF_{wa} [h]	f_{sw} [-]	f_{aw} [-]	$FF_{weighted}$ [h]	$FF_{weighted}$ [year]	EF [PAF.m^3/kg]	CFa [PAF.m^3.year/kg]
Atrazine	4461	3077	496	0.689	0.111	2690	0.307	131	40
Simazine	2606	1805	341	0.692	0.131	1585	0.181	47	9

Please note that in an LCA and comparative assessment context the overall impact score is calculated as the product of a characterization factor and the amount of chemical released into the environment (see Eq.1). This paper only focuses on modelling the so called characterization factors and do not address the inventory assessment, i.e. identifying how much of a pesticide is required for a similar application.

Conclusion

This paper demonstrated the feasibility of developing a comparative risk assessment based approach to identify chemical emissions having a high potential of affecting the aquatic ecosystem. The multimedia model IMPACT 2002, which combines a multimedia fate and an effect assessment model, has been used for this purpose on the Lake Geneva catchment. The fate model has been evaluated against monitored surface water concentrations of 22 pesticides used in agriculture. For half of the pesticides, modelled and monitored concentrations were found to be within the same range. For the other half, the concentrations were mainly underestimated. The differences were mainly explained by i) the supposed presence of other sources of pesticides than agriculture, such as 3 biocides used in urban areas and 5 pesticides manufactured by industries located in the water catchment and ii) by a poor estimation of degradation half-lives in water (generally underestimated). The ecotoxicological effect model adopted by IMPACT 2002, the AMI method, has been compared with the SSD method for two chemicals. Both methods yielded similar results.

Several limitations of this assessment warrant consideration. All data was collected from a limited number of farmers in the lemanic catchment and extrapolated to the whole region. It is therefore imprecise and a more robust model estimation could be achieved using more extensive datasets. Furthermore, the amounts of biocides and pesticides released from urban areas and industries are not known. Accounting for these emissions would allow a better understanding of the model estimation.

This application went beyond a sole evaluation exercise, and demonstrated how a modelling approach could help provide additional understanding in the complex issue of pesticide contamination of locally affected areas. The proposed comparative risk assessment framework enables to rank on an ordinal scale the potential risk of pesticide emissions and thus identify the most problematic ones. It can therefore be used as a management and decision making tool for farmer and local authorities, i.e. by promoting pesticides inducing little damage to the environment (for a similar application) or by restricting pesticides having the highest impact. This model can ultimately contribute in promoting the reasonable use of pesticides.

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Relating life cycle assessment indicators to gross value added for Dutch dairy farms

I.J.M. de Boer[1], M.A. Thomassen[2], and M.A. Dolman[3]
[1] Animal Production Systems Group, [2] Animal Sciences Group, [3] Agricultural Economics Research Institute, Wageningen University and research centre. Mark.Dolman@wur.nl

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Abstract

Sustainable dairy production requires farms that are economically viable, ecologically sound and socially acceptable. A good ecological performance of a dairy farm not necessarily implies a good economic or social performance. To gain insight into a possible “trade-off” between economic and ecological sustainability, we investigated in this study the relation between the ecological and economic performance of dairy farms, and their underlying characteristics. To determine such a relation, however, economic and ecological indicators are required for a relatively large number of dairy farms. The Agricultural Economics Research Institute in The Netherlands collects technical and economic figures from Dutch farms that subsequently are documented in FADN (Farm Accountancy Data Network). The economic and ecological performance of 119 specialized FADN dairy farms was assessed for the year 2005. Economic indicators used were gross value added per kg fat-and-protein-corrected milk (FPCM) and labour productivity. Ecological indicators used were: land use per kg FPCM, energy use per kg FPCM, global warming potential per kg FPCM, eutrophication and acidification potential per kg FPCM or per ha of land. Environmental indicators were based on an attributional LCA and economic allocation was used whenever a multifunctional process occurred.

Results showed that it was possible to perform an LCA for a large group of dairy farms based on FADN. Future LCAs based on FADN can be strengthened by extending FADN data collection with, for example, quantities of purchased products such as bedding material and seeds, mineral nitrogen content of purchased and produced manure, and information on soil content, such as phosphorus saturation. Results showed that farms with a high labour productivity (i.e., gross value added per total amount of labour) had a low on-farm energy use, total and on-farm land use, total and on-farm global warming potential, and total and off-farm acidification potential per kg FPCM . On the other hand, farms with a high labour productivity had a high on-farm eutrophication and acidification potential per hectare. From partial least squared regression analysis it was concluded that relations between economic and environmental were affected mainly by annual milk production per ha, annual milk production per cow, farm size, and amount of concentrates per 100 kg FPCM. Labour productivity, for example, increased as milk production per ha (results from Dutch livestock units per ha and annual milk production per cow) increased, which explained the relation between labour productivity and on-farm land use per kg FPCM. Similarly, the relation between labour productivity and global warming potential per kg FPCM could be explained partly by annual milk production per cow and kg concentrates/100 kg FPCM. The variation found in economic and ecological performance among farms shows that there is potential to improve economic and ecological sustainability. The fact that a high labour productivity relates to a low global environmental impact (energy use and climate change) but a high local environmental impact addresses the importance of balancing animal productivity and stocking density.

Introduction

The concept of sustainability was introduced to address concerns about our future livelihood (WCED, 1987). Sustainability is a holistic concept, consisting of three domains: economic, ecological and social, also referred to as the three pillars: profit, planet and people (Elkington, 1998). Most sustainability assessments of food production address only one of these thee domains. Many studies focus on environmental sustainability of agricultural production only, because environmental pollution is a side-effect of agricultural food production. Production of milk by dairy cattle, for example, contributes to nutrient enrichment of the ecosystem, climate change and acid deposition. Life Cycle Assessment (LCA) is used to evaluate the environmental impact of products throughout its life cycle (Guinée *et al*., 2002). Milk production by dairy cattle depends on many inputs, so the LCA method is justified to assess the environmental burden of milk production (Thomassen and de Boer, 2005; Dalgaard et al., 2006). An LCA of milk production gives us insight into the environmental domain of sustainability or the “planet” pillar. Preferably, however, more than one domain of sustainability should be addressed (Glavič and Lukman, 2007; Ness *et al*., 2007; Van Passel *et al*., 2007). Production of milk is not sustainable without economically viable farms, the pillar “profit” (Van Passel *et al*., 2004). An understanding of the relation between economic viability and environmental impact of milk production, therefore, is a prerequisite for a better insight into sustainability and to contribute to decision making (Norris, 2001; Mouron *et al*., 2006). To understand this relation, the relation between economic viability (i.e. economic performance) and environmental impact (i.e., environmental performance) of dairy farms needs to be assessed. Such an assessment requires a relatively large number of dairy farms. Most LCA studies of dairy cattle production systems, however, are based on a limited number of farms, because data collection is time-consuming (Cederberg, 1998; Cederberg and Flysjö, 2004; Casey and Holden, 2005; Thomassen *et al*., 2008a). Performing an LCA for a large number of farms enables us to differentiate results among farms and to study the relation between their environmental and economic performance and their underlying characteristics. The Farm Accountancy Data Network (FADN) of the Agricultural Research Institute in the Netherlands enabled us to perform an LCA and economic analysis of milk production for a large number of farms (FADN, 2007). The objective of this study, therefore, is to quantify the relation between the environmental and economic performance of FADN dairy farms, and to identify which farm characteristics influence this relation.

Material and methods

Farm Accountancy Data Network (FADN)

The economic and ecological performance of specialized FADN dairy farms were analyzed for 2005. The Agricultural Economics Research Institute in the Netherlands continuously collects technical and economic figures from Dutch farms that subsequently are documented in FADN. The objective of this documentation is to gain insight into the performance of a sector. In 2005, data of 271 dairy farms were collected, corresponding with the rate of appearance of the dairy farms in the Netherlands. As this study focuses on specialised conventional dairy farms, organic farms were excluded, and conventional farms were selected only when at least 75% of the economic size originated from dairy activity and no pigs and poultry were present. Due to a lack of indispensable data to perform an LCA (e.g., grazing system, milk urea content) or due to inconsistency of data (e.g., no specific data on purchased concentrates, while on the nutrient balance concentrates was given as an input), more farms were excluded from the analyses. In total, 119 dairy farms were analysed.

Ecological performance

The ecological performance of these 119 dairy farms was based on indicators derived from a Life Cycle Assessment (LCA). LCA is methodology that determines the environmental impact of all processes in the life cycle of an activity, in this case the production of milk (Guinée *et al*., 2002). Stages of an LCA include: goal and scope definition, inventory analysis, impact assessment and interpretation of results (ISO, 2006).

The goal and scope definition includes definition of the functional unit, the method of allocation and the system boundary. The functional unit chosen was 1 kg of fat-and-protein-corrected milk (FPCM) leaving the farm gate (CVB, 2004). This implies that the environmental impact is assessed for all process involved up to the moment that milk leaves the farm, i.e. production of purchased concentrates, roughage, bedding material, reared animals, manure, fossil fuels, fertilizer and pesticides. In addition, transport associated with production of purchased inputs was included.

Production of medicines, seeds and machinery were excluded because of their small impact (Cederberg, 1998). Buildings were excluded because we assumed similarity in buildings of different farms.

We performed an attributional LCA and used economic allocation to partition the environmental impact of a multifunctional process (Thomassen *et al*., 2008b). Multifunctional processes present were production of feed ingredients and bedding material, and production of milk, meat and manure at the dairy farm. Furthermore, the impact categories land use, energy use, acidification, eutrophication and climate change were assessed. These impact categories are important to consider when performing a cradle-to-farm-gate LCA of dairy farms (Berlin, 2002; Høgaas Eide, 2002; Thomassen *et al*., 2008a). Unlike land use, energy use and climate change, acidification and eutrophication were expressed per kg FPCM and per hectare, as these categories have a local and regional impact.

The inventory analysis consists of the collection of inputs, outputs and emissions related to each production process incorporated in the analysed system. In general, the same approach as presented in Thomassen *et al*. (2008a) was used (see Tab. 2 in article), adjusted to new insights, or adjusted to the way data were available in FADN, as described in the following paragraphs. Characterisation factors used for eutrophication and acidification were based on Heijungs *et al*. (1992), while characterisation factors used for climate change were based on IPCC (2006).

Production of concentrates and roughage

Purchased feed was divided into three categories: roughage, wet by-products and concentrates, based on the division made in the Dutch feeding value table (CVB, 2004). This division is based on dry matter content, besides practical insight of the feed industry. For each rough fodder, wet by-product and, singular concentrates, a Life Cycle Inventory (LCI) was computed, based on crop cultivation, crop processing and transport (Dolman, 2007). Crude protein content was used to distinguish among compound concentrates with different ingredients, and subsequently different environmental burdens (LCIs). Five types of concentrates were identified based on crude protein content, while in Thomassen *et al*. (2008a) three types of concentrates were identified based on intestine digestible protein content. Composition of each concentrates was based on annual data (>95% of its main feed ingredients) (Doppenberg and de Groot, 2005). Palm kernel expeller contributed for 15-20% to all five concentrates. Citrus pulp contributed for around 10% and soy hulls or wheat hulls for around 15% to the two concentrates with a low crude protein content (crude protein content <160 g/kg). Maize gluten meal contributed for around 25-30% and rape seed meal for around 15% to the three concentrates with a high crude protein content (160 g/kg< crude protein content <180 g/kg).

Purchased milk powder was included, based on an LCA of milk from conventional dairy farms supplemented with milk processing data of the dairy industry (Oldenhof, 2004; Thomassen *et al*., 2008a). Seeds (grass, rye, maize, potato, sugar beet and wheat) purchased by the dairy farm were included in the assessment, whereas seeds required for production of purchased feed were not included, because of lack of data (EcoinventCentre, 2004).

Production of milk

Thomassen *et al*. (2008a) used a fixed value to estimate methane emission from enteric fermentation. Preferably, a farm-specific emission rate must be used to include variation among farms. In this study methane emission from enteric fermentation was estimated by taking into account consumed feed types (e.g., concentrates ingredients, roughage, wet by-products). Smink *et al*. (2003) estimated emission factors (expressed in g methane/kg dry matter) of different feed types based on the fermentation of carbohydrates into volatile fatty acids (VFAs). The quantity of consumed feed per dairy cow was estimated taking into account energy demand for maintenance and production, production of grass and other crops at the dairy farm, purchased concentrates, and purchased other feed (wet by-products and roughage). In this study methane emission from enteric fermentation was computed by combining the methane emission factor per feed type and the quantity of this feed type consumed by the dairy cow.

Furthermore, to include variation among farms, besides nitrogen excretion, also ammonia emission during housing was related to farm-specific milk urea content based on Smits *et al*. (2003; 2005).

In addition, the way manure was applied to the field was known for each farm, which enabled to relate ammonia emission to the technique of manure application. Soil type was taking into account when estimating the amount of nitrate leached, and when estimating the amount of direct nitrous oxide emitted from agricultural land (Schröder *et al*., 2005; Schils *et al*., 2006; Schils *et al*., 2007).

Data assumptions

The following assumptions related to FADN were made to enable performing an LCA of the individual dairy farms (based on the LCA dairy farm model described in Thomassen *et al*., 2008a). No data on purchased quantities of sawdust were available, only costs. The cost price in 2005 of €0.16/kg sawdust was used to convert costs to quantities (Zevenbergen, 2006). The manure application technique was reported in frequencies, e.g., 40% injection and 40% narrow band spreading, without distinguishing between land types. Narrow band spreading is possible only on grassland (Van der Hoek, 2002), and therefore, this frequency was ascribed to grassland. Surface spreading in combination with ploughing only is possible on arable land (Van der Hoek, 2002), and therefore, this frequency was ascribed to arable land. The division of manure injection was made based upon the ratio grassland/arable land. No data were available on the mineral nitrogen content of purchased and produced manure, and therefore, a fixed value of 48% for semi-liquid and of 23% for solid manure were used (Mooij, 1996). Furthermore, we assumed that Dutch soils were saturated with phosphorus and, therefore, the total phosphate surplus was assumed to leach into the environment (Oenema *et al*., 2005).

Economic performance

Economic sustainable agriculture creates added value which is sufficient to remunerate all resources in an adequate way, both today and in the future “ (Van Passel *et al*., 2004). The economic performance of a dairy farm, therefore, was measured by computing its gross value added (Van Passel *et al*., 2004). The gross value added (GVA) is the difference between value of total production and non-factors costs (Barry *et al*., 2000). To correct for differences in scale among farms, GVA was expressed per kg of FPCM and per total amount of farm labour (i.e. both paid and unpaid labour). The GVA per unit of labour also is referred to as labour productivity (Van Passel *et al*., 2004).

Relation between ecological and economic indicators

Relations between ecological and economic indicators were quantified by a correlation analysis. Data were first tested for normality; the Pearson correlation test was used in case of normality, whereas the Spearman Rho’s correlation test was used in case of non-normality. We refer to a trade-off when a good economic performance (e.g., high labour productivity) was associated with a bad environmental performance (e.g. high global warming potential per kg FPCM) or the other way around.

To further explain the relations between ecological and economic indicators, we performed a Partial Least Squares (PLS) regression analysis. A PLS was used to estimate the correlation between two dependent variables based on a linear combination of orthogonal factors extracted from a group of independent variables. Dependent variables analyzed were the economic and ecological indicator for each significant correlation. Independent variables included in PLS were farm characteristics such as milk production per dairy cow, milk production per hectare, milk quota, farm size, Dutch livestock units per hectare, amount of purchased concentrates per 100 kg FPCM, amount of purchased roughage and wet by-products fed per 100 kg FPCM, diesel use per 100 kg FPCM, electricity use per 100 kg FPCM, gas use per 100 kg FPCM, milk urea content, purchased artificial fertiliser (kg N/ha and kg P_2O_5/ha), purchased animal manure (kg N/ha and kg P_2O_5/ha) and grazing system (division based on grazing hours). For each farm characteristic, a PLS analysis yields loading values for each extracted orthogonal factor. These loading values were used to describe which farm characteristics had an effect on the correlation analysed.

Results

General farm characteristics

Tab. 1 contains the general farm characteristics of 119 analyzed dairy farms. On average, these farms owned 53.4 ha of land of which 74% was grassland and 26% arable land and had a milk quotum of about 697 ton kg. The average number of cows was 85 with an annual milk production of around 8150 kg FPCM, resulting in an average production intensity of 12.5 ton FPCM per ha and 2 GVE per ha (Dutch livestock units).

Tab. 1: Mean and standard deviation of general farm characteristic of 119 dairy farms based on the Dutch Farm Accountancy Data Network in 2005.

Parameters	Units	Total
Farms	n	119
Grassland	ha	39.3 (22.4)
Arable land	Ha	14.1 (16.2)
Milk quota	ton FPCM[a]	696.8 (449.4)
Milking cows	N	85 (50)
Milk production	kg FPCM/cow	8151 (1214)
	ton FPCM/ha	12.5 (4.4)
Milk fat	%	4.41 (0.2)
Milk protein	%	3.51 (0.1)
Milk urea content	mg/100 gram	24.2 (3.5)
Stocking density	GVE[b]/ha	2.0 (0.5)
Purchased concentrates	kg/100 kg FPCM	23.1 (7.9)
Purchased other feed	kg DM[c]/100 kg FPCM	8.5 (9.9)
Diesel use	l/100 kg FPCM	1.0 (0.5)
Electricity use	kWh/100 kg FPCM	5.0 (2.1)
Gas use	m^3/100 kg FPCM	0.2 (0.3)
Artificial fertiliser	kg N/ha	146 (47)
Purchased animal manure	kg N/ha	30 (39)

[a-]FPCM is fact-protein-corrected milk.

[b]Dutch Livestock Units; 1 LU is annual phosphorus excretion of one milking cow.

[c] Dry Matter uptake by roughage and wet by-products.

Environmental and economic performance

Tab. 2 shows the mean and standard deviation of ecological and economic indicators for 119 specialized farms in 2005. Total land use was about 1.3 m^2/kg FPCM of which 54% was on-farm land use, 22% consisted of land use related to purchased concentrates and 19% of land use related to purchased roughage, by-products and bedding material. Total energy use was 5.3 MJ/kg FPCM of which 56% consisted of purchased concentrates and 16% was related to on farm energy use. Total climate change was 1.4 kg CO_2-eq/kg FPCM, of which 40% consisted of emissions related to keeping animals (mostly methane and nitrous oxide) and 24% of emissions related to purchased concentrates. Total eutrophication was 0.12 kg NO_3-eq/kg FPCM of which 58% consisted of on-farm ammonia and leaching of nitrate and phosphate, whereas 21% was related to purchased concentrates and 13% to purchased roughage, wet by-products and bedding material. Total eutrophication expressed per hectare, was 976 kg NO_3-eq/total hectare. Total acidification was 11.2 g SO_2-eq/kg FPCM of which 35% consisted of emissions related to keeping animals and 24% of ammonia emission related to fertilizer application, whereas 25% consisted of emissions related to purchased concentrates. Total acidification expressed per hectare, was 95 kg SO_2-eq/total hectare. The average gross value added per kg of FPCM was €0.28, whereas labour productivity equaled €112,000 per human year

Tab. 2: Environmental and economic performance of 119 dairy farms in 2005

Indicator	Unit		Mean (standard deviation)
Land use	m^2/kg FPCM	On farm	0.70 (0.2)
		Off farm	0.58 (0.3)
		Total	1.28 (0.4)
Energy use	MJ/kg FPCM	On farm	0.87 (0.3)
		Off farm	4.40 (1.3)
		Total	5.30 (1.3)
Climate change	kg CO_2-eq/kg FPCM	On farm	0.76 (0.1)
		Off farm	0.61 (0.2)
		Total	1.36 (0.3)
Eutrophication	kg NO_3-eq/kg FPCM	On farm	0.07 (0.03)
		Off farm	0.05 (0.02)
		Total	0.12 (0.04)
	kg NO_3-eq/on farm ha	On farm	1025 (563)
	kg NO_3-eq/off farm ha	Off farm	907 (169)
	kg NO_3-eq/total farm ha	Total	976 (334)
Acidification	g SO_2-eq/kg FPCM	On farm	7.1 (2.0)
		Off farm	4.2 (1.4)
		Total	11.2 (2.6)
	kg SO_2-eq/on farm ha	On farm	101 (26)
	kg SO_2-eq/off farm ha	Off farm	85 (32)
	kg SO_2-eq/total farm ha	Total	95 (19)
Gross value added	€/kg FPCM		0.28 (0.05)
Labour productivity	K €/human years		112 (55)

Relating environmental and economic performance

Correlations between economic and ecological indicators are in Tab. 3. Results show that a high on-farm land use per kg FPCM (extensive farms) was associated with a high GVA per kg FPCM (r = 0.36; P<0.001). From PLS, we concluded that milk production per ha and diesel use per FPCM explained this correlation. Farms with a low milk production per ha had a relatively high feed production on farm (high diesel use per kg FPCM). In case of a high feed production on farm, the amount of purchased concentrates was low. Consequently, total feed costs were low and therefore GVA per kg FPCM was high. The negative correlation between labour productivity and on-farm land use (r = -0.33; P<0.001) was affected mainly by milk production per ha and farm size. Literature shows that labour productivity is higher as farm size increases. Furthermore, this correlation implies that labour is more efficiently used on intensive farms because labour efficiency is higher for milk than for feed production.

Furthermore, farms with a high labour productivity have a low total global warming potential per kg FPCM (r = -0.27; P<0.01) and a low amount of greenhouse gas emission at farm level per kg FPCM (r = -0.26; P<0.01). Farms with a high labour productivity have a low on-farm feed production, and therefore a low emission of N_2O from on-farm application of fertilizer. In addition, farms with a high labour productivity have a higher milk production per ha and generally have cows with a higher annual milk production. A high annual milk production per cow is associated with a low CH_4 emission per kg of milk due to dilution of CH_4 emission related maintenance.

Tab. 3 also shows that a trade-off was observed (positive correlation) between labour productivity and area-related indicators such as eutrophication and acidification potential per ha. Farms with a high labour productivity have a high eutrophication potential per hectare of farm land (mainly nitrate and phosphate leaching) (r = 0.21; P<0.05). In addition, farms with a high labour productivity have a high acidification potential per hectare (mainly ammonia emission) of farm land (r = 0.25; P<0.01). These trade-offs are due to the fact that farms with a high labour productivity have a high milk production per ha (reciprocal of on-farm land use per kg FPCM) because of a high stocking density and a high annual milk production per cow.

Tab. 3: Correlation between economic indicators, i.e., gross value added (GVA) per kg FPCM and labour productivity, and Life Cycle Assessment (LCA) indicators of 119 dairy farms in 2005.

LCA indicators[a]	*Unit*	*GVA/kg FPCM* r^a	*Labour productivity* r^a
Total Land use	m^2/kg FPCM	ns[c]	-0.26**
On farm Land use		0.364***	-0.33***
Off farm Land use		Ns	Ns
Total Energy use	MJ/kg FPCM	Ns	-0.27**
On farm Energy use		0.299***	-0.21*
Off farm Energy use		Ns	Ns
Total Global Warming Potential	kg CO_2-eq/kg FPCM	Ns	-0.27**
On farm Global Warming		0.306***	-0.26**
Off farm Global Warming		Ns	Ns
Total Eutrophication Potential	kg NO_3-eq/kg FPCM	Ns	Ns
On farm Eutrophication Potential		Ns	Ns
Off farm Eutrophication Potential		Ns	Ns
Total Eutrophication Potential	kg NO_3-eq/total farm ha	Ns	Ns
On farm Eutrophication Potential	kg NO_3-eq/on farm ha	Ns	0.21*
Off farm Eutrophication Potential	kg NO_3-eq/off farm ha	Ns	Ns
Total Acidification Potential	g SO_2-eq/kg FPCM	0.250***	-0.22***
On farm Acidification Potential		Ns	ns
Off farm Acidification Potential		0.369***	-0.21*
Total Acidification Potential	kg SO_2-eq/total farm ha	Ns	Ns
On farm Acidification Potential	kg SO_2-eq/on farm ha	Ns	0.25**
Off farm Acidification Potential	kg SO_2-eq/off farm ha	Ns	-0.22*

[a] r= Spearman Rho's correlation. ns= not significant; *P<0.05; **P<0.01; ***P<0.001

Discussion

The results of the life cycle analysis are within the scope of former research (Cederberg and Flysjö, 2004; Thomassen *et al.*, 2008a). Former LCA studies of dairy cattle production systems analyzed a limited number of farms (20-30 at highest), whereas this study used FADN data to perform an LCA of 119 dairy farms. Future LCAs can be strengthened by the inclusion of more suitable LCA-related data within FADN. Only specialized conventional dairy farms were included in the analysis. The farms that were analyzed had a larger farm size (53.4) than an average Dutch dairy farm (42.3), and a higher number of milking cows (85) than an average Dutch dairy farm (65) (LEI, 2007). Annual milk production per cow and milk production per ha, however, were similar to the average Dutch farm (respectively 8150 FPCM per cow and 12.5 ton FPCM per ha (LEI, 2007). Furthermore, labour productivity was slightly higher than an average Dutch dairy farm (96,1 k€) (LEI, 2007). This is because farms analyzed in this study were of larger size, which generally results in a higher labour productivity because of more efficient use of labour.

The objective of this paper was to quantify and explain the relations between ecological and economic performance of dairy farms. Farms with a high labour productivity had a low on-farm energy use, total and on-farm land use, total and on-farm global warming potential and total and off-farm acidification potential per kg FPCM . These indicators were product-related and expressed per kg FPCM. On the other hand, farms with a high labour productivity had a high on-farm eutrophication and acidification potential per ha. These indicators were area-related and expressed per hectare. Milk production per cow influenced all LCA indicators expressed per kg FPCM, as amount of milk produced is the denominator of these indicators. Stocking density (livestock units per ha) influenced all LCA indicators expressed per hectare. Other studies also showed choice of functional unit influences LCA outcomes (Van der Werf *et al.*, 2007; Thomassen *et al.*, 2008a).

Conclusion

This study demonstrated that farms with a high labour productivity had a low on-farm energy use, total and on-farm land use, total and on-farm global warming potential and total and off-farm acidification potential per kg FPCM . On the other hand, farms with a high labour productivity had a high on-farm eutrophication and acidification potential expressed per hectare. Farm characteristics that influenced these relationships between environmental and economic performance were: milk production per hectare, annual milk production per cow, farm size and purchased concentrates per 100 kg FPCM. The variation found in economic and ecological performance among farms shows that there is potential to improve economic and ecological sustainability. The fact that a high labour productivity relates to a low global environmental impact (energy use and climate change) but a high local environmental impact addressed the importance of balancing animal productivity and stocking density.

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Developing a Methodology to Integrate Private and External Costs and Application to Beef Production

R. Teixeira[1], C. Fiúza[1], T.Domingos[1]
[1] Instituto Superior Técnico, Environment and Energy Section, DEM, Av. Rovisco Pais, 1, 1049-001 Lisbon, Portugal, ricardo.teixeira@ist.utl.pt

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Abstract

In this paper we develop a method for the integration of Life Cycle Assessment (LCA) and economic valuation. The main application of this method is trade-off analysis between environmental and economic performances. We use an aggregate indicator for the environmental performance, namely Eco-Indicator99 (EI99) from the LCA software SimaPro.

EI99 measures human health in DALY – Disability Adjusted Life Years. We use conventional economic valuation of the Value of Statistical Life (VSL) to obtain 74-175 k€/DALY. The quality of this estimate is confirmed by back-converting from DALY to GHG emissions, using the conversion factors in SimaPro, obtaining a valuation range of 16-37 €/ton CO_2. We then use the weight for human health in EI99 to obtain an economic valuation for EI99 points: 2.83€-6.71€/point. This is the monetary valuation for the other impact categories.

Using this method, we compare beef production in two extensive animal production systems: (1) natural poor grasslands, and (2) sown biodiverse permanent grasslands. Contrary to general belief, we conclude that the latter, although more intensive, are better, and would be even more so if their use of phosphate fertiliser were optimised. Private costs are higher in the natural pastures scenario due to the greater area needs, and this is mainly reflected by the fencing costs. Sown pastures have low private costs because they are more productive. Hence, costs per steer are smaller.

Introduction

With some limitations, cost-benefit analyses are an easy way to determine trade-offs between environmental services or impacts and private costs of a product or activity, since both are expressed to economic terms.

Life Cycle Assessment (LCA) is probably the most complete method for determining environmental impacts, since they incorporate all impacts originated by a product, from resource extraction to waste disposal (Goedkoop, 1998). In principle, it is based on a careful and holistic accounting of all energy and material flows associated with a system or process. It is used to compare the environmental impacts associated with different products that perform similar functions (Mayerhofer *et al.*, 1997). LCA has been used in European studies such as Labouze *et al.* (2003) or Tukker *et al.* (2005), to identify the products or product groups with greatest environmental impact from a life cycle perspective. LCA can even go as far as to produce a single number, aggregating all environmental impacts, through indicators such as Eco-Indicator99.

However, LCA is restricted to material and energy flows, disregarding economic information (Kytzia *et al.*, 2004), which means that it disregards the use of capital and labour. Economics provides approaches to this problem, through the use of economic valuation methods. These methods allow the conversion of environmental data to economic values, which can then be added to conventional market-based private costs. An example of the application of economic valuation methods is the ExternE project, launched by the European Commission in the 90's, to provide a scientific basis for the quantification of energy related externalities. The value of a statistical life (VSL) was used for valuing fatal accidents and mortality impacts in climate change modelling (Mayerhofer *et al.*, 1997). A more recent update of ExternE (Friedrich, 2004) comprises, for example, a valuation of environmental impacts using the standard price approach (use of the abatement costs of emissions reductions as a proxy for the revealed willingness of European society to pay for the improvements in ecosystems health) to obtain shadow prices for global warming (5 to 22 €/ton CO_2).

In this paper, we propose a method to assess private costs and value the total environmental impacts and services of a given activity by carrying out an LCA analysis and then valuing these impacts economically. This work is divided in two parts: method definition, and application to a case study.

Regarding the definition of the method, it starts with an LCA performed with Eco-indicator 99 (EI99), using the software SimaPro 7.0. Luo *et al.* (2001) mention EI99 as one of the major environmental impact assessment methods, comprehensive in nature and generating a single numerical value reflecting the composite magnitude of global impact associated with a specific product. Toffel and Marshall (2004), focusing on toxic release data, recommend it, among 13 leading weighting methods, for the analysis of impacts on human health and the environment.

EI99 is based on a damage-oriented methodology, focusing on three types of environmental damages damage to Human Health, expressed as Disability Adjusted Life Years (DALY); damage to Ecosystem Quality, expressed as the loss of species over a given area, during a certain time; and damage to Resources, expressed as the surplus energy needed for future extraction of minerals and fossil fuels. Since EI99 adds the values in these impact categories, it is possible to use economic values for one of them to obtain a valuation for the others. This is very easy to do with DALY's, because they can be easily measured with economic valuation methods for the value of statistical life.

As for the application of the method, we used beef production as a case study. Namely, we compare two scenarios: production using natural pastures; production using sown permanent biodiverse pastures (in the sense of Teixeira *et al.*, 2007). The choice of scenarios is due to qualitative evidence that sown pastures, although requiring a larger amount of inputs, provide multiple environmental services, which may offset the impacts from a larger amount of required inputs. In fact, sown biodiverse pastures allow a higher sustainable stocking rate, by increasing soil organic matter, a key factor for water retention, erosion decrease and carbon sequestration. Nitrogen fixation by legumes reduces the consumption of nitrogen fertilizers, the production of which has a high energy cost with correspondent high greenhouse gases emissions. Raising the stocking rate and reducing nitrogen fertilizer consumption leads to increased economic viability of the farm. Mediterranean sown pastures are also carbon dioxide sinks. The average absorption potential is estimated as 5 ton CO_2/ha.year during the first 10 years (Teixeira *et al.*, 2007). On average, carbon sequestration in natural pastures does not occur because tillage is periodically necessary for the removal of shrubs, fire prevention and pasture maintenance; this tillage event leads to the degradation of any organic matter accumulated in the preceding years.

But in both cases the animals still require some level of feed supplementation during the less productive seasons. Therefore, we performed an LCA of several types of commercial feed in order to choose the one that minimizes total environmental impact. We studied all steps in feed production, namely ingredient production, industrial processing and transportation.

Method / Approach

Life Cycle Assessment

The method we defined starts by determining environmental impacts through LCA, namely using the EI99. EI99 extends and updates the Eco-indicator 95 methodology, developed by the National Reuse of Waste Research Program and by Pré Consultants of the Netherlands. Impact assessment is done to obtain a single numerical value (the Single score total). The following steps are normalization and weighing:

$$\text{Single score} = \frac{\text{IC}}{\text{NV}} \times \text{WF}, \tag{1}$$

where IC is the impact category or the damage category, NV is the normalization value (a reference value for Europe) and WF is the weighing factor. We used the default weighing set, which corresponds to the "Hierarchist" version of the EI99 methodology, with scientifically and politically accepted underlying value choices (Goedkoop and Spriensma, 2001). Tab. 1 contains the NV and WF factors for the three damage categories.

Tab. 1: Normalization and weighing factors for EI99.

Damage category	Normalization	Weighing
Human Health	1.54E-02 DALY/yr	400
Ecosystem Quality	5.13E+03 PDF.m^2.yr/yr	400
Resources	8.40E+03 MJ/yr	200

By estimating an economic value for the DALY, we can account for damages assessed by EI99: we convert the €/DALY value into €/single score point (Pt) by using Equation 1 and values from Tab. 2. This way, we convert environmental impacts (aggregated and for each theme) to monetary units.

From environmental impacts to monetary units

DALY is an indicator used by organisations such as the World Bank and the World Health Organisation as a tool to allocate money to health care, to assist in setting health service priorities, to identify disadvantaged groups and to provide a comparable measure of output for intervention, program and sector evaluation and planning (Homedes, 1996). It measures the total amount of ill health, due to disability (YLD: Years Lived Disabled) and premature death (YLL: Years of Life Lost), attributable to specific diseases and/or injuries. It is based on a disability weighing scale, between 0 (perfectly healthy) and 1 (death).

There is considerable variation in the assigned monetary value among studies that quantify the value of disability and lost life. It is most commonly calculated using estimates of the quality of life, Willingness To Pay (WTP) for safety measures, wage premia for risky jobs and individual behaviour related to safety measures, reflecting individuals' risk valuations, either elicited directly through surveys or revealed in their labour market decisions (Kenkel, 2002).

Labour market studies, upon which the VSL estimates are usually drawn, measure compensation for risk of instantaneous death for people about 40 years old and thus value approximately 35 years of life (Lvovsky, 1998). An approach to estimate the Value of a Statistical Life Year (VSLY) is to regard it as the annuity which when discounted over the remaining life span of the individual at risk would equal the estimate of VSL (Pearce, 2000):

$$\mathrm{VSLY} = \frac{\mathrm{VSL}}{[1-(1+r)^{-n}]/r}, \tag{2}$$

where n is the expected life remaining (35 years) and r is the consumption discount rate (4% by EC Guideline, according to Friedrich (2004)).

Having reviewed more than 60 studies of mortality risk premia from ten countries, Viscusi and Aldy (2003) found a great deal of heterogeneity in VSL estimates. We consider as adequate the range of VSL put forth by them, US$5 million to US$12 million (2000$), the median being US$7 million. Given the absence of studies for Portugal, we will adopt this range of values, bearing in mind that transferring unadjusted estimates is clearly hazardous (although widely practised), due to differences in the socio-economic characteristics of the relevant populations, in the physical characteristics of the study and policy site.

So, to update the values mentioned above, we used the inflation rate for the USA of 16.59%, obtained for the period between January 2000 and December 2005 (Inflation Calculator, 2006). Income is best adjusted for through Purchasing Power Parity (PPP), given the uncertainty about the variation in WTP across populations of different incomes, although further research needs to be done on its income elasticity (EC, 2000). According to World Bank (2005), the PPP for Portugal is $19,250 while for the US it is $39,710 (2004' values), which gives a 0.48 factor. Currency exchange has been around 1.2 US$/€ (Banco de Portugal, 2006). Our range of VSL thus becomes 2.0 - 4.8 M€. The value of 3.1

MECU (1995) obtained by Martins *et al.* (1998) fits in this range (given the inflation rates in the European Union in the period 1995-2005). VSLY, using Equation 2, ranges from 78 k€ to 186 k€. Assuming this to be the value of a DALY, we convert it to €/single score point with Equation 1, obtaining the range 3.0 €/Pt to 7.2 €/Pt.

A thorough analysis of each case is imperative. Not only due to the possible inadequacy of some of the EI99's underlying data but also due to some externalities specific to the case in question. For these unaccounted impacts, the solution is to use the customary environmental valuation techniques. An example is the use of a Replacement Cost Method for soil loss presented latter on in the paper.

Application to a case study

Our case study focuses on beef production in extensive systems in either sown or natural pastures in Portugal. Our choice was not only influenced by the fact that food from animal origin is mentioned as having a big environmental impact (Labouze *et al.*, 2003), but also by the fact that carbon uptake by grasslands has been chosen by the Portuguese Government to fulfil its Kyoto Protocol's target.

Characterization of the system

In this part of the study, the functional unit we found more appropriate was animal produced so that we can quantify the impacts to produce a 12 month old "ready to slay" calf. Natural and sown pastures allow different stocking rates. The average stocking rate for sown pastures is 1.18 CU/ha, and for natural pastures 0.44 CU/ha (Carneiro *et al.*, 2005). We consider the steer's mother is equal to 1 CU, whilst the steer is equal to 0.6 CU.

Based on Domingos *et al.* (2005), we developed a setting in which the beef calf is fed only on maternal milk until the age of 7.2 months. Also, the cow gives birth on a yearly basis, so we take into account a year of its life and emissions (12/12×1CU). We considered that the calf's emissions are not relevant until after 7.2 months of age, when it grazes for 2.4 months, its diet consisting of 40 % pasture (2.4/12×0.6CU×0.4) and 60 % (2.4/12×0.6CU×0.6) industrially processed feed. From 9.6 until 12 months old (2.4 months), the calf is kept in a stable, fed only on industrially processed feed and silage maize (2.4/12×0.6CU), until taken to the slaughterhouse. Adding up, we have, per year, a CU equivalent of 1.048 for pasture and for the feed a CU equivalent of 0.192. To obtain the equivalent area needs per year per steer for each type of pasture, we divide the CU equivalent for pasture with the average CU/ha. We therefore have for sown pastures 0.89 ha/steer.yr and for natural pastures 2.38 ha/steer.yr. The average steer weighs 1.9E+02 kg when 7.2 months old and by the time it reaches 12 months, it weighs 3.6E+02 kg. For all scenarios, the amount of industrially processed feed is the same, adding up to 1.1E+03 kg in the 4.8 months it is fed to the steer and it.

With data from crop fact sheets in GPPAA (2001) and others constructed by us, based on data collected from Quinta da França, a farm situated in Cova da Beira (Central Portugal), we simulate the production systems in each case by multiplying the areas above for each type of pasture, thus comprising all impacts from every input (fertilizers, chemicals…) and every action (sowing, harrowing…). While simulating sown pastures, additional considerations had to be made concerning distances of transportation to Quinta da França. On the other hand, data for gaseous emissions from cattle, pasture and manure (discriminated in the following sections) need only to be multiplied by the CU value corresponding to pasture time or feed time (considered to be spent in the stable). Data regarding the implementation of sown pastures is to be divided by ten, because re-sowing may occur every ten years. An additional scenario was considered, for "mature" sown pastures, i.e., self-sustainable pastures where nutrients run in a closed cycle. Although no longer functioning as carbon sinks, as they have reached maximum soil organic carbon, they have no need for human interference.

Greenhouse gas emissions from pastures and cattle

Even though sources of N_2O are poorly quantified, microbial processes in the soil are believed to be by far the greatest source. Two dissimilar energy-yielding microbial processes generate N_2O production in soil: nitrification (predominantly regulated by ammonium availability) and denitrification (regulated by the availability of nitrate, oxygen, and reduced forms of carbon). The substrate for each of these processes is determined by the relative rates of N mineralization and N assimilation by plants and microbes and by diffusion constraints (Mummey *et al.*, 2000)

In natural pastures, a model developed by Mummey *et al.* (2000) allowed to obtain N_2O emission values for grasslands in the USA, which were coherent with values from previous studies. The mean flux for all regions was 0.28 kg N_2O-N/ha.yr. Since no other studies were found for natural pastures, we used this value. On the other hand, in sown pastures, legumes are responsible for N fixation and for N_2O emissions. According to Ledgard (2001), gaseous losses from grazed, temperate legume/grass pastures range from 0.5 to 5 kg N_2O-N/ha while Rochette and Janzen (2005) obtained, for non fertilized annual crops, an emission value of 1.0 kg N_2O-N/ha, which we chose to use.

Methane emissions are originated by cattle through enteric fermentation and by anaerobic decomposition of manure. When deposited on pastures, manure tends to decompose aerobically therefore producing little or no methane. Typical methane and nitrogen emissions are presented in Tab. 2 (IPCC, 1997; PNAC, 2003).

Tab. 2: Beef cattle methane emissions and nitrogen emission data for manure.

Methane emissions (kg/CU)	Enteric Fermentation	48
	Manure (from stable)	1.88
Manure Nitrogen (kg N/CU)	Pasture	21.6
	Manure from stable placed on soil	32.4
N_2O Emission Factors (kg N_2O/kg N excreted)	Pasture	0.02
	Manure from stable placed on soil	0.0125

Emission factors are different probably because, as stated in IPCC (1997), the amount of N_2O released depends on the system and duration of waste management; as fresh dung and slurry are highly anoxic and well buffered with near neutral pH, one would expect N_2O production to increase with increasing aeration, which initiates the nitrification-denitrification reactions, allowing the release of N_2O.

According to IPCC (1997), for cattle, faecal excretion is usually about 8 g N/kg dry matter consumed, regardless of the nitrogen content of the feed, the remainder being excreted in urine. The bulk of the N_2O will be lost shortly after deposition in the field (up to 50 % of the mineral nitrogen in animal manure, i.e., about 25 % of total N). This may be the reason why nitrogen content in faeces in pastures is lower than in manure from the stable; in the latter it is not as easily volatilised and the N content of the feed is higher than that obtained by grazing.

According to Flessa *et al.* (2002), losses of N_2O–N from dung heaps are about 0.1–0.8% of the manure N. The values presented by PNAC (2003) are within this interval. Direct N_2O emissions from cattle were not included, as these were considered to be negligible (Flessa *et al.*, 2002).

Greenhouse gas emissions from lime application

Liming is responsible for the emission of CO_2, obtained by multiplying the annual amount of calcic limestone (tons/year) by an emission factor of 0.12 (IPCC, 2003).

Leaching and runoffs

The only fertilizer applied is phosphate. Its runoff value is 0.01 kg PO_4/kg P input from synthetic and organic fertilizer (van der Werf *et al.*, 2005). We only consider surface runoff losses since, according to Turner and Haygarth (2000), the transfer of P through subsurface pathways is, in agronomic terms, of minor importance, due to the large capacity for P fixation in the usually P-deficient subsoil, with P export representing <1 % of the applied fertilizer and a minute fraction of the total soil P.

Additional impacts

In order to add water consumption as an impact, since it is not considered in LCA software SimaPro, we can estimate how much water is available for the crop with irrigation. An economic value for water can be adopted from the price imposed by the Portuguese government for water for agricultural uses from the Alqueva dam in Alentejo, a major investment in the driest area of Portugal. From the year 2008 on, the price is 0.08 €/m^3 and so we will consider it as the highest value.

On the other hand, a Water Resources Tax (TRH) is being developed by the government, which aims to reflect the economic value of the good, internalising costs due to degradation of water and investments on water resources, as an incentive for a more sustainable use, in the context of the Water Framework Directive. Here we adopt a simplification of it, concerning only costs from using water from the Public Hydric Domain and diffuse pollution emissions:

$$\mathrm{TRH}=\mathrm{V}\times\mathrm{trh_u}\times\mathrm{C_{sect}}\times\mathrm{C_{scar}}\times\mathrm{C_{efic}}+\sum_{j=1}^{k}\mathrm{trh}_{\mathrm{d}_j}\times\mathrm{M}_j \qquad (3)$$

Where V is the used volume of water (m^3), $\mathrm{trh_u}$ is the unit tax for the use (€/m^3), C_{sect} is the sector coefficient (0.2 for agriculture), C_{scar} is the scarcity coefficient (1.15 for the site's hydrographical basin), C_{efic} is the efficiency coefficient (0.65 for agriculture), $\mathrm{trh_{dj}}$ is the unit tax for diffuse pollution over each indicator of potential contamination j (0.03 €/kg of N and P emitted) and M_j is the potentially contaminating quantity for indicator j (kg).

To assess the cost of soil loss, the method used by Marta *et al.* (2005) for the Zonal Program of Castro Verde (Portugal) was adopted. Although soil erosion can virtually disrupt all the environmental services provided by soil, due to data constraints, only the cost of replacing soil productivity was estimated, with a Replacement Cost Method. Using the cost of replacing nutrient (organic matter, P and K) losses and the cost of returning to farmland the eroded sediment, we have:

$$\mathrm{RC}=(\mathrm{S}_t-\mathrm{S}_{t+1})\left[\sum_{j=1}^{k}\mathrm{N}_j\mathrm{P}_j+\frac{\mathrm{P_r}}{\mathrm{B_d}}\right] \qquad (4)$$

Where RC is the replacement cost of nutrients and eroded sediment removal (€/ha), S_t - S_{t+1} is the soil loss from time t to t+1 (ton/ha), N_j is the quantity of the jth nutrient in the soil (kg/ton), P_j is the price of the jth nutrient (€ /kg), B_d is soil bulk density (1.5 ton/m^3) and P_r is the market price of dredging 1 m^3 of sediment (€/m^3).

Results

LCA results

As we are modelling a steer, the amount of time confined in a stable and the share of industrially processed feed consumed is the same for all scenarios. Note that to obtain a 12 month-old steer, we need pastures for 1 year plus 2.4 months and only 4.8 months of feed, which may minimize its impacts. As may be seen in Tab. 3, they may even reach values below those for natural pastures. The impacts where the feed clearly surpasses pastures are: Acidification/Eutrophication, mainly due to soybeans (due to its fertilizer) and, as it is the feed's major component, of the silage maize (due to its fertilization process) and Land Use. Also relevant are Fossil Fuels and Ozone Layer (due to transportation).

Tab. 3: Eco-indicator 99 characterization results and direct space used for sown and natural pastures and feed.

Impact category	Unit	Feed	Natural pasture	Sown pasture	Sown pasture (11th year)
Carcinogens	DALY	6.50E-05	1.10E-05	1.10E-04	-
Resp. Organics	DALY	3.60E-07	7.00E-07	1.00E-06	6.40E-07
Resp. Inorganics	DALY	3.80E-04	5.10E-05	4.40E-04	-
Climate Change	DALY	9.60E-05	3.00E-04	-5.50E-04	3.10E-04
Radiation	DALY	7.20E-07	8.70E-08	1.70E-06	-
Ozone Layer	DALY	3.50E-08	4.00E-09	3.20E-08	-
Ecotoxicity	PAF*m2yr	1.20E+02	2.80E+01	2.40E+02	-
Acidification/ Eutrophication	PDF*m2yr	3.30E+01	1.50E+01	1.20E+01	-
Land Use	PDF*m2yr	2.40E+01	0.724	7.70E+00	-
Direct Space Use	ha	9.00E-02	2.40E+00	8.90E-01	8.90E-01
Minerals	MJ surplus	8.10E+00	3.60E+00	2.20E+01	-
Fossil Fuels	MJ surplus	4.30E+02	5.00E+01	3.70E+02	-

We also performed an uncertainty analysis for single score results in EI99. Results for the two types of pasture plus feed were obtained through a Monte Carlo analysis with SimaPro 6.0, generating random numbers to determine the parameters in the uncertainty domain of all values in the database used. Results for error comparison with both cases show that, in 98% of the cases, simulated within the parameters' error margins, the single score impact for natural pastures is lower.

Private costs

Yearly costs per steer can be divided in implementation and maintenance costs. Since they are fully invested in the first year, we convert implementation costs to constant annuities for the time horizon of the product or service as follows:

$$\mathrm{P}=\sum_{i=1}^{n}\frac{\mathrm{A}}{(1+\mathrm{t})^{i}}=\mathrm{A}\,\frac{1\text{-}(1+\mathrm{t})^{-n}}{1\text{-}(1+\mathrm{t})^{-1}} \tag{5}$$

Where P is the total amount paid in the first year (€), A is the annuity value (€), t is the interest rate (1.5%) and n is the time horizon (years). Implementation of sown pastures is considered to be necessary every ten years. Fencing costs 280 €/ha and lasts for ten years. The steer's mother costs 750€ and we consider it to live and breed for 15 years.

Tab. 4: Private costs for the farmer. General costs include labour, machinery costs and general expenditure, rent, interest on circulating capital.

		Natural pasture €/steer	Sown Pasture €/steer	Sown pasture (11th year) €/steer
Implementation	General costs	-	35.68	-
	Fence	67.09	25.09	25.09
	Breeding Cow	50.53	50.53	50.53
Maintenance	General costs	129.45	117.22	3.10
	Feed	105.74	105.74	105.74
	Silage maize	19.08	19.08	19.08
	Labour with cattle	80.68	80.68	80.68

Water use and soil loss costs

Since pastures are rain fed, we only focus on the industrially processed feed, quantities and origin specified by Teixeira *et al.* (2005), as shown in Tab. 5.

Tab. 5: Irrigation needs for the industrially processed feed's ingredients.

Location[4]	Ingredient	Quantity (kg)	Irrigation water (m^3/ton ingred.)	Irrigation water (m^3/steer)
ALE	Maize (silage)	635.94	9.50E+01	6.00E+01
RO	Maize (grain)	135.65	6.00E+02	8.10E+01
ARG	Soy (44% protein)	98.76	1.40E+03	1.40E+02
ALE,RO	Wheat (grain)	71.62	1.10E+02	8.20E+00
U.S.A.	Corn Gluten Feed	71.62	4.80E+01	3.40E+00
ALE,RO	Wheat (straw)	62.94	0	0
			Total	**2.90E+02**

Using the value of 0.08 €/m^3, we obtained a total cost of 23 €/steer. In order to apply the Water Resources Tax (Equation 3), we need to estimate the contamination from N and P, so we multiplied the total fertilizer inputs of 9.37 kg N and 5.86 kg P (Teixeira *et al.*, 2005) by the emission factors 0.02 kg NH_3-N/kg N applied and 0.01 kg PO_4-P/kg P applied, taken from van der Werf *et al.* (2004). We obtain a total value of 0.6 €/steer.

According to Crespo (2004), in 10 years, a legume rich pasture increases its organic matter (OM) content from 1% to 3%, corresponding to a rate of 0.2%/year. Soil loss from pastures was estimated with Wischmeier's Universal Soil Loss Equation (USLE). The soil is mainly sandy, so an OM content of 1% (natural pastures) represents a soil loss of 1.329 ton/ha.yr while for 3% OM ("mature" sown pastures) it is 0.208 ton/ha.yr. As for the "young" sown pasture, we chose the 3 year-old scenario, with an OM content of 1.6%, corresponding to a 0.993 ton/ha.yr loss.

Total costs

Adding all the previous contributions, we obtain the total results shown in Tab. 6.

[4] ALE, RO - Alentejo, Ribatejo e Oeste (Portuguese regions). ARG - Argentina.

Tab. 6: Values in €/steer.yr for all components considered.

	Natural pasture + Feed		Sown pasture + Feed		Sown pasture (11th year) + Feed	
	Min	***Max***	***Min***	***Max***	***Min***	***Max***
Water	0.6	23	0.6	23	0.6	23
Soil	5.82		4.96		1.34	
Carcinogens	5.88	14.11	13.65	32.77	5.04	12.1
Climate Change	31.32	75.17	-35.67	-85.6	32.07	76.96
Respiratory Inorganics	33.84	81.22	63.75	153.01	29.85	71.65
Other LCA categories	17.39	41.73	26.76	64.24	16.2	38.89
Total external costs	**94.85**	**241.05**	**74.06**	**192.37**	**85.09**	**223.94**
Private costs	**452.27**		**434.02**		**284.22**	
Total	**547.12**	**693.32**	**508.08**	**626.39**	**369.31**	**508.16**

Discussion

LCA results

One would expect results to be worse for sown pastures, given the amount of inputs needed when compared to the almost inexistent ones for natural pastures. This clearly underlines the need for assessing all impacts deriving from any activity or for comparison purposes. We must bear in mind that the area needed for raising a steer in sown pastures is a third of the one for natural pastures. As for the "mature" sown pastures, impacts are almost non-existent, as expected, except for the climate change category. This is mostly because of the CH_4 emissions from cattle (60% of the climate change category). The remaining 40% are due to the high N_2O emission factor from legume grasslands, an effect concealed in the younger sown pastures as they act as carbon sinks. While in natural pastures harrowing is the main contributor for negative impacts, in sown pastures it is mainly the phosphate applied. Considering Climate Change, sown pastures are clearly preferable since they act as carbon dioxide sinks and therefore they have negative impacts.

Land Use measures the damage as a result of conversion or occupation of land and has nothing to do with land occupied by pastures (Direct Space Use), which is greater in the case of natural pastures. So, Land Use has a higher weight for sown pastures, mainly due to the phosphate production and transportation used (shed to store equipment, etc). In the case of the Feed, the direct space occupation is very low because its main ingredient, the silage maize, has a productivity of about 42 tons/ha while its Land Use is very high due to the seeds needed and their production and transportation processes.

Ecotoxicity appears to have a significant weight for both types of pasture, as seen in Tab. 3. This may be explained due to substances such as heavy metals, which are not usually considered when analysing agricultural life cycles. As for the Respiratory Inorganics category, in sown pastures it is mainly due to the phosphate applied (from the sulphuric acid needed for its production and transportation means involved) and the tillage rotary cultivator and tractor (machinery's emission from combustion, tyre abrasion and fuel) while for natural pastures it is mainly due to harrowing (fuel and machinery).

Total costs

Though the lower €/DALY value is a third of the higher, this is concealed by the magnitude of the private costs compared to the costs determined by LCA. Private costs are higher in the natural pastures scenario due to the greater area needs, and this is mainly reflected by the fencing costs. Sown pastures have such low private costs because they are more productive. Hence, costs per steer are smaller.

Also, to compare costs for environmental impacts in the Climate Change category with those from other studies we used the EI99's damage values in DALY per kg of CO_2. Despite the differences in the underlying methods, we obtained a range of 16.6 - 39.4 €/ton CO_2 eq., which falls near the range of 5 to 22 € per ton of CO_2, provided by NewExt (Friedrich, 2004), and the EU Emissions Trading Scheme value of 26 €/ton CO_2 (Katoomba, 2006).

Conclusion

This paper aims to develop a method for integrating Life Cycle Assessment with economic analysis and to apply this method to a case study: beef production on natural pastures vs. sown pastures. Moreover, aggregating into an economic value is quite a simplified output, understandable by either a scholar or by a layman and aiding in cost-benefit analysis. As can be seen in Tab. 6, to produce a steer in sown or natural pastures is somewhat the same cost if we only account for private costs (although natural pastures have very low maintenance inputs, the greater area requirements make them more expensive). However, the gap between them deepens once we include LCA results. Also, “mature” sown pastures are even less costly, so in the long run sown pastures prove to be even better.

Using this type of LCA tool has inherent problems as well as limiting assumptions, such as: emissions, land uses and all subsequent damages are considered to occur in Europe; there are various error sources (statistics, extrapolation, expressing impacts in DALY); capital goods and auxiliary products are usually required but frequently, given system boundaries set for the analysis, they are not taken into account. There are also limitations regarding the EI99. In the economic assessment, the lack of data and adequate studies for Portugal imposed that we undertake “alternative” paths in order to fulfil our goal, like the emission factors adopted or the chosen methods for pricing water. Also, in the assessment of soil loss costs, assuming soil loss is represented by productivity loss is quite a simplification. It is important to point out that, as results are affected the same way by the inherent errors and bias sources, we do not expect variations in parameters to change the conclusions. A good indicator that our method is trustworthy is the fact that the economic value range obtained for impact per ton of CO_2 is consistent with those obtained in other studies.

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Using LCA data for agri-environmental policy analysis at sector level

Schader C. [1], Nemecek, T.[2], Gaillard, G.[2], Sanders, J.[1,3] and Stolze, M.[1]
[1] Institute of Organic Agriculture (FiBL), Frick, Switzerland
[2] Agroscope Reckenholz-Tänikon (ART), Zürich, Switzerland
[3] Johann-Heinrich von Thünen Insitute (vTI), Braunschweig, Germany
E-Mail: christian.schader@fibl.org

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Abstract

In times of limited agricultural budgets, the cost-effectiveness of the policies becomes a major decision criterion for policy reforms. For comprehensively assessing the effects of an agri-environmental policy environmental data has to be scaled up to sector level, taking into account uptake rates and transaction costs of the policies. This paper discusses the general suitability of LCA data to be upscaled and combined with economic sector models. We present an approach that is based on the representative farm-group model FARMIS and the Swiss Agricultural Life Cycle Assessments (SALCA).

Exemplary results of the model for energy use are shown and most prominent issues in the context of this upscaling process are discussed. The paper argues that uptake-effect functions will not necessarily be linear. Furthermore, the combination of normative and positive datasets causes inconsistencies which need to be minimised. Finally, we argue differences between the life-cycle view and the economist's perspective lead to difficulties in the interpretation of the results. Nevertheless, this approach may deliver plausible results and can supplement ecological site-specific studies in the evaluation of agri-environmental policies.

Introduction

The Swiss agricultural policy has been implementing a progressive environmental agenda since the introduction of direct payments in 1993. Full cross-compliance was introduced already in 1998 and additional ecological services were stimulated by targeted agri-environmental payments, including payments for organic farm management. Against the background of a limited budget, the considerations on cost-effectiveness play a fundamental role for a further development of the direct payment system (Badertscher, 2004). Therefore, from a policy-maker's perspective, it is essential to have reliable data about costs and effects of single policy measures or policy mixes.

Up to now, studies analysing environmental effects at sector-level are relatively scarce, though there are recent efforts from the scientific and the policy side to bridge this gap using economic models (Britz and Heckelei, 2008). However, to link these models to environmental indicators is delicate because environmental effects are difficult to aggregate and requires reliable environmental data that have been generated in a consistent framework. The life-cycle assessment approach is one of the most widespread approaches, and in Switzerland, agricultural production has been analysed extensively (Nemecek *et al.*, 2005).

Therefore, this paper pursues the following objectives:

- To present an approach linking life-cycle assessment data to an economic sector model
- To explore the suitability of life-cycle assessment data for policy analysis at agricultural sector level based on conceptual considerations and exemplary calculations with the model

- To highlight essential issues in the upscaling of LCA data to sector level

In this paper, firstly, the specific methodological approach used in this study is presented. Secondly, the upscaling process is demonstrated by calculating energy use at agricultural sector-level based on LCA data. Thirdly, the main challenges of the upscaling procedure are raised and the applicability of such a model discussed.

Modelling approach and conceptual considerations

In this section, we firstly delineate the general principles of the economic sector model FARMIS and then describe the way LCA data is linked to the model.

The economic sector model FARMIS

Our analytical approach consists of an economic sector model linked with life-cycle assessment data. The economic model FARMIS is a sector-consistent static-comparative farm group model, which can be used for the assessment of policy impacts at sector level. The model is primarily based on farm accountancy data from the Swiss FADN distinguishing between 29 plant production activities and 15 animal production activities (Sanders *et al.*, 2008).

Employing Positive Mathematical Programming (PMP) (Howitt, 1995), FARMIS optimises the objective function (1) under consideration of restrictions that express the limitations in economic resources such as land, labour and capital as well as political restrictions such as the Swiss cross-compliance regulation.

$$\begin{aligned}\max Z_n = & \sum_j \sum_k p_{njk} Y_{njk} - \sum_i \sum_k c_{nik} X_{nik} + \sum_i \sum_k dp_{nik} PX_{nik} - \sum_u r_{nu} U_{nu} - \\ & \sum_v r_{nv} V_{nv} - \sum_l r_{nl} LAND_{nl} - \sum_i \sum_k \delta_{nik} X_{nik} - 0.5 \sum_i \sum_k \omega_{ni1} X^2_{nik} - 0.5 \sum_i \sum_w \omega_{ni2} X^2_{niw} \quad \forall n\end{aligned} \tag{1}$$

$Y_{njk}, X_{ni}, PX_{ni}, U_{nu}, V_{nv}, LAND_{nl} \geq 0$

where:

<u>Indices</u>:

n = index for farm groups
i = index for production activities
j = index for output products
k = index for intensity levels
w = index for intensity levels ≠ w
l = index for land type
u = index for labour
v = index for fertilisers

<u>Parameters</u>:

p = prices for agricultural products
c = activity-specific costs
dp = activity-specific direct payments
r = variable costs
δ = parameter for linear hidden cost
ω = parameters for quadratic hidden cost (depending on the alternative intensity levels)

<u>Variables</u>:

Z = objective (profit per farm group)
Y = sales of agricultural products
X = level of activities
PX = level of activities eligible for direct payments
U = level of labour input/requirements
V = level of fertiliser input/requirement
LAND = level of rented UAA

The cost-effectiveness of policy measures on sector level can be assessed quantitatively with this economic modelling approach taking into account **uptake rates**, **environmental effects**, and the **public expenditure** as three major determinants of a successful agri-environmental measure. In this paper we focus on the question of measure environmental effects at sector level using LCA data.

Determination of environmental effects at agricultural sector level

The most frequently studied issue about agri-environmental policies is their effectiveness in achieving policy objectives, i.e. minimisation of negative environmental impacts of agriculture (e.g. Stolze *et al.*, 2000; Bengtsson *et al.*, 2005). There are different types of environmental impact assessment used, one of the most relevant approaches is the ISO-standardised life-cycle assessment approach (Wood, 2003).

In Switzerland, extensive life-cycle assessments of agricultural activities (Swiss Agricultural Life-Cycle Assessments (SALCA)) have been carried out (Nemecek *et al.*, 2005) supplemented by data from the ecoinvent database (Frischknecht *et al.*, 2007). SALCA data has been calculated for the most relevant impacts of agricultural activities that are typical for Swiss agriculture. Data for farming activities is differentiated by farming system (integrated and organic farming), region (valley, hill and mountain region) providing a sufficient detailed basis for the model analysis in Switzerland. Furthermore, the environmental impacts of the most important agri-environmental measures are covered. Of the possible impact categories, direct and indirect energy use, nitrogen and phosphorus eutrophication and species biodiversity have been integrated as impact indicators for each activity and management intensity in FARMIS.

There are both direct, i.e. on-farm use of primary energy, and indirect energy use components, i.e. inputs for agricultural production, which themselves require the input of primary energy for their production in agriculture. For the modelling of **energy use**, we based our analysis on ecoinvent and SALCA data (Nemecek *et al.* 2005). Additional data was gathered for activities that were not explicitly covered by SALCA or ecoinvent. Both direct (i.e. fuel, gas, electricity) and indirect energy use (i.e. seeds, plant protection, fertiliser, feedstuffs, machines, buildings) were modelled.

Within CH-FARMIS, there is a normative link to the SALCA **eutrophication** data. As the basis of the SALCA eutrophication data, nitrogen and phosphorus models calculate eutrophication potential in dependence of key factors like season and types of application (Prasuhn, 2006; Richner *et al.* 2006). Simultaneously, FARMIS calculates nutrient balances, independent of seasonal differences of application, according to the fertiliser purchase of farm groups, based on FADN data. The model allows a comparison between the results of the eutrophication potential and the nutrient balance. The parallel usage of a pressure and a state indicator for eutrophication allows mutual comparison and verification of the results of both procedures.

Besides eutrophication effects, **biodiversity** effects belong to the most studied environmental impacts of agriculture (e.g. Bengtsson *et al.*, 2005). As there are the general relations of management practices and intensity of agricultural practices (Faucheux and Noël, 1995), there is a principal possibility to take into account biodiversity impact within aggregated economic models without referring to detailed site-specific characteristics (Mattison and Norris 2005).

The SALCA biodiversity indicators express the habitat quality for 11 groups of species. Groups with high ecological requirements (i.e. amphibians, locusts, butterflies, spiders and carabid beetles) obtain a special emphasis in the biodiversity model. Further, groups of indicator species are flora on arable land, flora on grassland, birds, small mammals, molluscs, butterflies, bees and locusts. The value for total biodiversity expresses a weighed mean of all groups, with weightings according to their specific importance in the food chain of a habitat, as proposed by Jeanneret *et al.* (2006). The biodiversity model considers the most important species-specific impacts of agricultural crop cultivation practices. This allows for a detailed coverage of the impacts of agricultural policies on species level at macro-scale.

Exemplary calculations of energy use at agricultural sector-level

On the basis of the methodical descriptions of the approach, the average energy use per ha has been calculated for the base year 2000/01. The calibration procedure of FARMIS ensures that the area covered by different crops exactly matches the real situation in the base year. In the calculations, dairy farms (FAT-Type 21) (Meier, 2005), beef farms (FAT-Type 22 and 23), and mixed farms (FAT-Type 51, 52, 53, and 54) were considered (Fig. 1).

The presented calculations include the indirect energy use in seeds, plant protection, fertilisation, and buildings (for machines and stables) and direct energy use such as on-farm machinery use and other on-farm processes. In-stable processes like feeding, milking and removal of manure are not considered in the presented results. Therefore, compared to similar studies (Mack *et al.*, 2007), the total figures seem to underestimate the average energy use per ha by about 10-30%, depending on the farm group. Furthermore, the differences between organic and conventional farm group are potentially underestimated, because purchased fodder has not been taken into account.

Fig 1 shows that the main share of energy use lies within sowing and harvesting (including transport and drying) as well as in farm buildings (including stables). For conventional farms, also fertilisation is responsible for a major share of energy use. Seeds are negligible for dairy and beef farms but have a visible share on mixed farms. Tillage also contributes more to energy use on mixed farms than on dairy or beef farms.

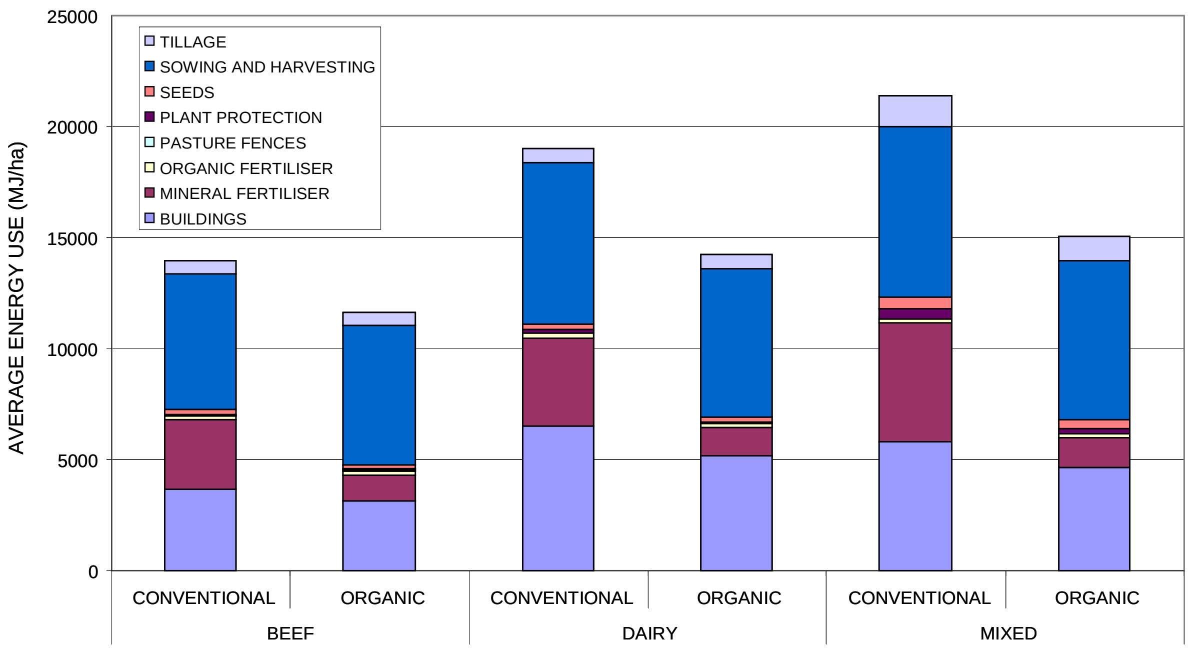

Fig. 1: Average energy use per ha in the base year differentiated by farm type and farming system

Besides the estimation of average sector values, shares of different farming systems and regions on the energy use at sector level can be calculated. So, farming systems, farm types and regions can be compared by means of representative data. Moreover, reactions of the different farm types to agri-environmental policy can be modelled.

The environmental indicators can be related to almost all financial indicators, which are calculated by the model, e.g. the agricultural income or sector-level added value.

Discussion

In this section we delineate what we find the key points for upscaling the environmental effects to sector level using LCA data.

Upscaling the environmental effects to sector level

In most of the studies, the environmental effects of farming types are studied at field or farm level. Only few studies (e.g. Julius *et al.*, 2003; Schmidt and Osterburg, 2005; Pufahl, 2007) conceptually combine the effects of the agri-environmental policies at farm level with the achieved uptake, which necessarily has to be done in order to analyse the sector level effects of policies. The basic issue for upscaling from field or farm level to sector level is whether a linear relation between uptake rates and effects can be assumed. The potential reasons for non-linearity, i.e. decreasing, increasing or variable marginal effects at sector level can be of different nature:

• Deadweight effects and self-selection bias: Deadweight effects occur for the first hectares under a policy because there is empirical evidence that those farms take up a policy where there is no or almost no change in management necessary (Henning and Michalek, 2008).

• Regional differences and differences between farm types: an agri-environmental measure will have a larger impact, if it is implemented on a specialised cash crop farm than on an already extensively managed mixed farm (Pufahl, 2007).

• 1st Gossen Law (law of decreasing marginal utility): The more of a good is consumed, the lower the gains in utility are. Although this law is developed for commodities, the relationship can be observed also for non-commodities. For example, the utility of a further decrease in nitrate content in drinking water may be high, if the content exceeds a set threshold, but it may be low, if the level of nitrate is already low (Schader *et al.*, 2007).

• Minimum ecological requirements: contrary to the 1st Gossen Law, there might also be cases in which marginal utility increases with higher uptake. Sometimes, a minimum of landscape complexity must be reached, before any additional positive effect on species biodiversity can be achieved due to the uptake of agri-environmental measures. Although this effect is locally specific, it can be argued that it leads to a different effect curve at sector level (Roschewitz *et al.*, 2005).

Possible relations between uptake (U) and cumulative environmental effects (E) are shown in Fig. 2. The marginal environmental effect at sector level ($\frac{\partial E}{\partial U}$) may be constant, increasing, variable or decreasing. The shape of the curve is different for different environmental objectives and indicators. Due to data constraints, the exact course of the uptake-effect curve cannot be observed empirically. However, using econometric models the curves can be estimated, provided that individual farm data on the environmental impacts is available (Frondel and Schmidt, 2005).

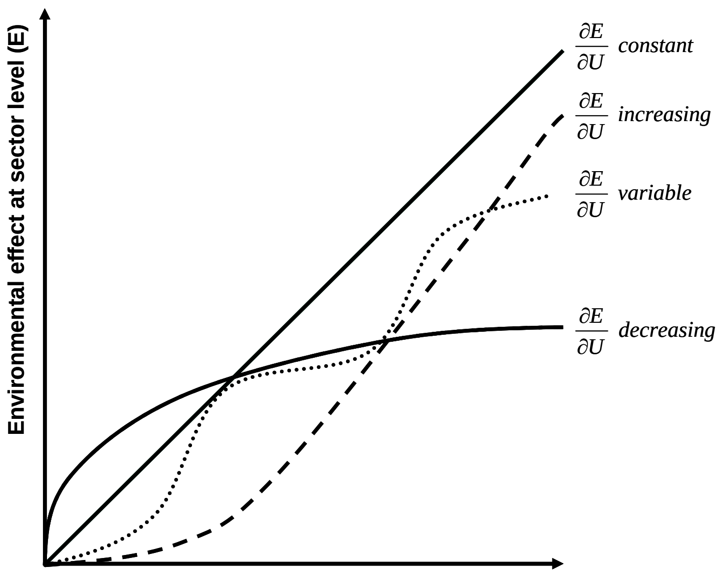

Fig. 2: Environmental effects of an agri-environmental policy in relation to its uptake rate (own representation).

Conceptual discrepancy between the LCA approach and the economic perspective

In sector-level calculations, we face a conceptual difficulty due to the combination of different system borders: In economic calculations, e.g. for input-output tables, monetary flows are described between sectors. Each sector is accounted for what comes in from other sectors and what goes out to other sectors. The LCA approach, however, has different system borders: Because it is product-based, all inputs and processes that were necessary to produce a certain product are charged, regardless of whether these processes occur in the agricultural sector or not. This means, for instance, that the production of one ton of wheat the production of mineral fertilisers and pesticides used have to be taken into account from a life cycle perspective, whereas in economic calculations production of fertilisers would be clearly allocated to the respective industry sector. This problem becomes even more sophisticated, if we include supranational monetary flows, which are becoming more important also in the agricultural sector (e.g. imported soybean-based fodder from Brazil). We argue that it is plausible to combine these two views for the purpose of agri-environmental policy evaluation, although a purely macro-economic perspective cannot be taken.

Linking normative and positive datasets

Despite the, in general, similar structure of the data sets of FARMIS and SALCA, data consistency problems occur when bringing together the two tools. Both FARMIS and SALCA consist of various datasets that are partly of strictly positive nature, and partly supplemented with standard data. FARMIS employs a positive approach, claiming to represent the reality “as it is”, therefore fertilisation, for instance, is calculated in FARMIS via FADN data, that is the actual expenses of a sample of representative farms for fertilisers. On the basis of fertiliser prizes and fertiliser needs for different crops, FARMIS calculates farm-group specific fertiliser uses.

Linked to SALCA data, which in the case of fertilisation normatively assumes, applications according fertiliser recommendations, we get two different assumptions of fertiliser use for the economic calculations on the one hand and for the ecological impacts on the other. The only way to generate consistency would be to completely decide to opt either for the normative or the positive way. Opting for the completely normative way means giving up the positive character of the model, which is a major advantage of FARMIS. At the same time, opting for the completely positive way is not feasible because for some data components, normative data simply are the more realistic data source due to the absence of positive data in the FADN dataset. For instance, the number of machines used for different crops may be more realistically estimated using purely normative assumptions than on the respective FADN accounts (e.g. depreciation). Therefore, since both ways have their disadvantages, it makes sense to compromise and neglect dataset consistency at the same time. We argue that for each component of each environmental indicator the data source with the least uncertainties has to be opted for. However, this procedure requires to transparently point out the assumptions behind the models and to validate the calculations within a transdisciplinary team of involved scientists.

Conclusions

The preliminary results presented above demonstrate the general feasibility of using LCA data within sector models. Combining LCA data with economic calculations at sector level leads to plausible results for the indicator energy use. As highlighted above, however, integrating other indicators in the model leads to conceptual difficulties. Effects on biodiversity, for instance, may not be upscaled to sector level linearly without introducing further assumptions. We found another major problem is the different setting of system boundaries in economic and environmental analyses. Furthermore, the linkage of different datasets partly of normative, partly of positive nature requires making further assumptions.

Against the background of the highlighted problems we conclude that using LCA data for agri-environmental policy analysis at sector level requires:

- different implementation procedures for each indicator, as the upscaling behaviour may follow a linear or a non-linear curve.

- the combination of multiple datasets and the minimisation of inconsistencies between them by harmonising assumptions
- particular transparency regarding the assumptions made and the system boundaries drawn, as economists and environmental scientists' have differing ways of drawing boundaries

Despite these requirements, the described approach allows a comprehensive evaluation of the modelled agri-environmental policies. Not only the effects of single agri-environmental policies, but also interactions between policies and policy mixes can be modelled and analysed. Thus, we are confident the presented approach will deliver plausible results and can supplement ecological site-specific studies in the evaluation of agri-environmental policies.

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Sustainability Solution Space for the Swiss milk value added chain: Combing LCA data with socio-economic indicators

C.R. Binder[1], J. Steinberger[1,2], H. Schmidt[1], A. Schmid[1]
1: Social and Industrial Ecology, Department of Geography, University of Zürich, Winterthurerstrasse 190, CH-8057 Zürich, Switzerland
2: Institute of Social Ecology , Faculty for Interdisciplinary Studies, University of Klagenfurt, Vienna, Austria
e-mail address: claudia.binder@geo.uzh.ch

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Abstract
We present the Sustainability Solution Space (SSP) an approach for assessing the sustainability of the Swiss milk value added chain. Current integrative and indicator-based assessment approaches in agriculture usually have three main shortcomings: (i) there is an overall focus on assessing the ecological aspects of agriculture neglecting to some extent economic and social aspects; (ii) research has so far focused on filling important gaps in knowledge and technology, but has missed to include the step towards utilization and implementation of this knowledge; and (iii) the assessment results themselves are difficult to be implemented in decision-making, as conflicting goals and the interaction between indicators has not been sufficiently considered. We propose that for filling this gap an approach is needed which fulfils *systemic criteria*, i.e., sufficient representation of the system including functional interaction among indicators, which allows to depict goal conflicts; *normative criteria*, i.e., considering the different value perspectives of stakeholders by including them in the process and designing sustainability ranges rather then threshold values; and (iii) *procedural criteria*, i.e. pursuing the assessment in a true transdisciplinary process. We present the SSP and its application for the Swiss milk value added chain. The system is described with a set of 17 indicators, 8 ecological (derived from LCA data) and 9 socio-economic. The sustainability thresholds were obtained through literature research and stakeholder interviews. The relationship among the indicators was developed in a transdisciplinary workshop. The SSP program takes a geometric approach to determine the intersection space corresponding to the satisfaction of the normative ranges while taking into account the functional interactions of the indicators. We show some results of the sustainability solution space for the Swiss milk value added chain and discuss the prerequisites, advantages and shortcomings of the method.

Introduction
Swiss agriculture is in a state of flux, transitioning from a heavily-subsidized sector to an integration into the multilateral trade accord system (Bundesrat 2006). The Swiss constitution's article 104 states that the agriculture should help Switzerland be self-sufficient, clearly contrasting social and environmental issues (food security, countryside maintenance) with economic valuation. Since 1992, with the Agricultural Reform and increasingly with the bilateral agreement on agriculture passed with the EU in 1999, which came into force in 2002, Switzerland has moved away from market support of production prices and towards direct payments for services to farmers (BLW 2005; Jung 2006). During this time, the number of farmers, farms and total farmland area have been steadily decreasing (BLW 2005).

Swiss milk production represents a quarter of Swiss agricultural revenue. Like the rest of Swiss agriculture, the Swiss dairy sector has seen a decrease in almost every dimension: animals and farms, while animal productivity has continued to increase, leading to almost a constant total milk production. Thus, Swiss dairy production still meets more than the Swiss national demand, with significant exports of some dairy products (especially cheese, milk and yoghurt). Total imports represented 6% by weight of domestic dairy consumption, and exports accounted for 9% of domestic consumption. Thus the vast majority of Swiss dairy production is consumed domestically, with a slight overproduction for export.

Because of the risk of overproduction leading to falling prices, Swiss milk production has been subject to government-imposed quotas (Lüthi 2001). Since May 2006, individual exemptions to the quotas can be granted under certain conditions (BLW 2006b), as a preparation for the planned total suppression of quotas in 2009 (BLW 2005). The requirements quota-lifting are the joining of dairy farmers in organisations consisting of dairy farmers or farmers-transformers-retailers of dairy products. The overall quota remains, applied by the organisation (including the administratively costly tasks of data-reporting). During this transition to more collective and farmer-based decision-making an integrative sustainability assessment methodology is required, which provides some insights into the room for action for a sustainable development and allows for analyzing the potential effect of strategies and policies. The main question is: what strategies and policies are advisable for producer-transformer organisations in order to manage their production in a sustainable direction?

Indicator lists for assessing sustainability have been created at international, national and sectoral level (UN 1993; BUWAL and BFS 1999; Jesinghaus 1999; Jesinghaus and Montgomery 1999; Renn, Leon *et al*. 2000; UBA 2000; CSD 2001). Regarding indicator-based sustainability assessment in agriculture Girardin *et al*., 2000; Rigby *et al*., 2000; Woodhouse *et al*., 2000; van der Werf and Petit, 2002), the following weaknesses have been encountered (Binder *et al*., submitted):

- the multi-functionality in agriculture is often not specifically addressed in sustainability assessments (Rossing *et al.*, 2007);
- there is an imbalance in the modeling and assessment work performed regarding the three dimensions of sustainability, i.e., ecological, economic and social aspects, in favor of the ecological one (von Wirén-Lehr, 2001);
- research has so far focused on filling important gaps in knowledge and technology, but has omitted to include the step towards utilization and implementation of this knowledge (Rossing *et al.*, 2007); and
- the assessment results themselves are difficult to implement in decision-making, as conflicting goals and the interaction between indicators has not been sufficiently considered (Morse *et al.*, 2001).

In this paper, we present a tool, Sustainability Solution Space (SSP) to cope with part of these shortcomings. We apply the SSP for assessing the sustainability of the Swiss dairy value added chain. SSP combines methods from Industrial Ecology (MFA, LCA) and with transdisciplinary approaches. It consists of the following elements:

- a multifaceted, comprehensive indicator system, inclusive of the multi-functionality and multidimensionality of the agricultural system as well as the perspectives of the different stakeholders,
- an integrative system analysis, including the interaction among indicators and providing an understanding of the systemic role of the indicators
- a sustainability assessment, defining a solution space for sustainability by combining expert and practitioner knowledge.

Method / Approach

Sustainability Solution Space: The Method

The core components of the SSP procedure (Wiek and Binder, 2005) are described in Tab. 1. Preliminary to constructing an SSP the function the sustainability space has to fulfill has to be defined (*prerequisite phase*). Who will use this tool and for what purposes? The transdisciplinary approach in this prerequisite phase allows for including and balancing the different views and objectives stakeholders might have.

The method itself consists of a systemic, a normative and an integrative module (Tab. 1). The modules are interdependent; constructing an SSP is thus not a linear procedure but an iterative process.

The *system module* is the basis for the sustainability solution space. It (i) describes and defines the system with its characteristics and its main problems, (ii) derives indicators (environmental, economic and social), and (iii) determines the relationship among the indicators. Note that the system module is already constructed in a transdisciplinary process, i.e. with participation of stakeholders.

The *normative module* sets the criteria for defining sustainability ranges. It includes both the stakeholder as well as the scientific view. For each indicator a sustainability range is defined, i.e., a minimum and maximum value is set according to the selected criteria.

The *integrative module*, finally, integrates the normative module and the system module. With a computer tool (see below) the sustainability solution space is calculated. It shows within which ranges the values of the indicators are sustainable and allows for analyzing trade-offs of measures.

Tab. 1: Steps of SSP adapted to sustainability assessment of agriculture (after Wiek and Binder, 2005; Binder and Wiek, 2007; Binder *et al*. submitted)

Step	Description
Prerequisite	
	Goals setting
	Stakeholder involvement
	Scale
Module I: Systemic Module	
Step 1	Characterizing the region to be assessed
Step 2	Problem oriented derivation of indicators (e.g., ecological, economic and social)
Step 3	Analyzing the inter- and intralinkages among the indicators as well as their dynamics
Module II: Normative Module	
Step 4	Specifying the sustainability ranges for the indicators
Module III: Integrative Module	
Step 5	Defining the solution space for decision-making
Step 6	Analyzing trade-offs

Selection of Indicators

Ecological Indicators

The selection of the ecological indicators was based on the Life Cycle Approach. First the impact categories relevant for the milk value added chain were defined. Second, the indicators having the largest impact within these categories were identified and quantified (Schmidt, 2007). The criteria for the final selection of the indicators were: (i) relevance of environmental impact; (ii) dynamic development (trend analyses); (iii) data availability; (iv) simplicity and preciseness; and (v) comprehensibility (Fig. 1).

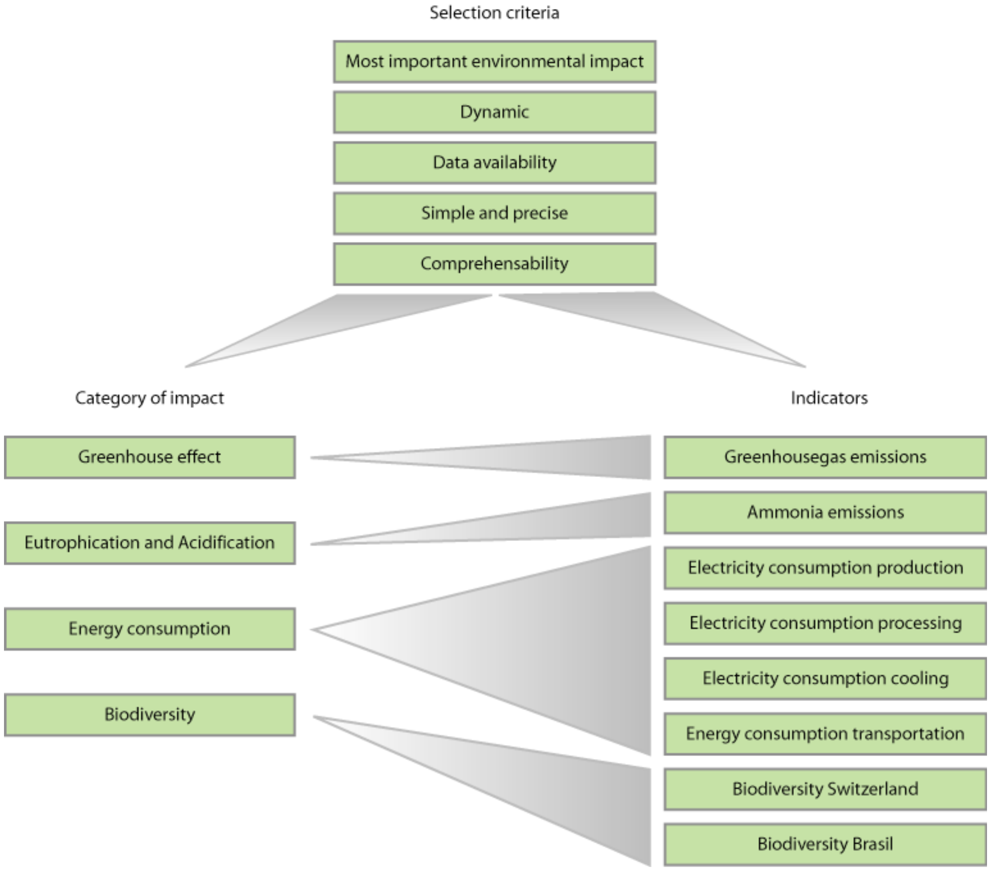

Fig. 1: Selection of the ecological indicators (after Schmidt 2007; Binder *et al.*, 2008)

Socio-economic indicators

For the selection of the social indicators both societal and individual aspects of the people working in the milk value added chain were considered (Schmid, 2008; Binder *et al.*, 2008). The economic indicators include macro-economic as well as micro-economic aspects (Fig. 2). Indicators such as hourly wage can be allocated to both, the social and the economic dimension of sustainability.

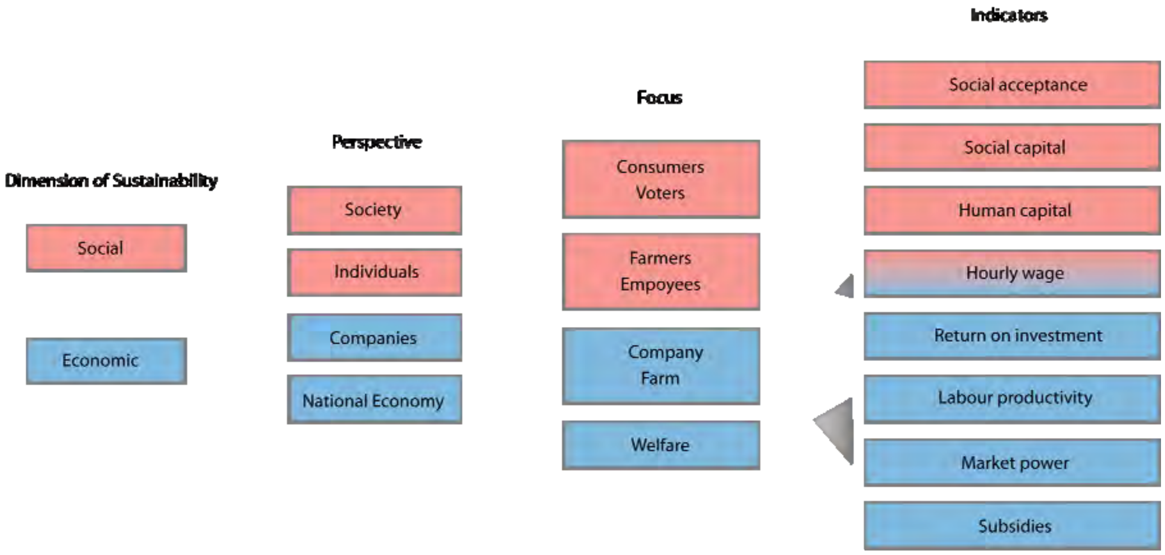

Fig. 2: Selection of the socio-economic indicators (Schmid, 2008; Binder *et al.*, 2008)

Definition of the solution space

To model the SSP, a geometric computer program, based in the Matlab language, was designed and implemented. This model takes an N-dimensional space of indicators, ranges and relations between the indicators, and finds the solution space corresponding to the intersection of the ranges and relations. Theoretically, N is unlimited, in practice, N should be under 15, since the computation becomes too numerically intensive (involving at least 2^{15} points in N-space).

The model starts with the indicators, along with their **upper and lower sustainability** ranges. For some indicators, only one of these limits may be relevant (since, for example, it is impossible to emit too few pollutants); in such cases, a theoretical upper or lower limit can be set. The first step is the computation of the trivial, initial sustainability space: the N-dimensional rectangle determined by the upper and lower range.

The next step integrates the **relations between the indicators**. These relations are functional boundaries between the indicators. These relations represent the dependency of one indicator upon another: for example, a certain emission level of pollutant emission may be permissible, but only if another pollutant's level is lowered. The sustainability space boundaries of these two pollutants are thus related to the value of the respective pollutant levels. The SSP computer program takes these value-dependent boundaries into account in calculating the final N-dimensional solution space. This process is shown in two dimensions in Fig. 3.

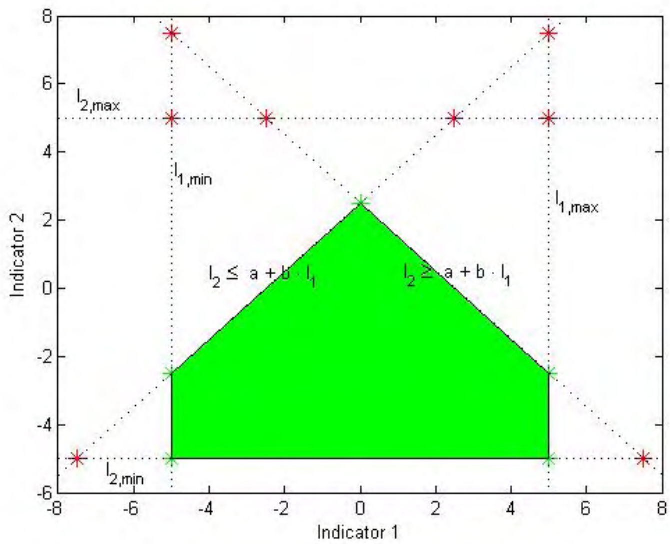

Fig. 3: Example of intersection points and resulting SSP in two dimensions.

Currently, the SSP model is limited to linear relations between the indicators, such as the oblique lines shown in Fig. 3. The possible results of the SSP model are the following: (1) an empty SSP; (2) the SSP is a unique point; (3) a line; (4) a 2-dimensional area; (5) a 3 to N-dimensional volume defined by its corner point coordinates. This space can then be used to identify potential pathways or policies towards a system state which respects the sustainability constraints of the individual indicators and their relations to one another.

Results

The selected indicators

Seventeen indicators were selected. Eight describe the ecological and nine the socio-economic dimension. They are presented in Tab. 2 and 3.

Tab. 2: Ecologic indicators

Indicator	Definition	Sustainability range	Desired development[1]	Unit
Greenhouse gas emissions	Greenhouse gas emissions (TGE) per kg of milk produced	0 – 0.76	-	TGE/kg milk
Eutrophication/ Acidification	Ammoniac emissions (NH_3) due to milk production	not used for SSP[2]	-	tNH_3/year
Electricity cons. milk production	Electricity required for producing of 1kg of milk	200 - 400	-	kWh/ kg milk
Electricity cons. processing	Electricity consumption for processing milk to milk products	not used for SSP	-	MJ/kg milk
Electricity cons. cooling	Electricity required for cooling milk products	not used for SSP	-	MJ/ kg milk
Energy cons. transportation	Energy required for transporting 1kg of milk	not used for SSP	-	MJ/ kg milk
Biodiversity Switzerland	Area of extensively managed pasture in the lowlands per kg milk	33'659 - 41'110	+	ha/ kg milk
Biodiversity Brazil	Loss of rainforest area due to fodder production for milk production	0 – 0.00025	+	ha/ kg milk

Tab. 3: Socio-economic indicators

Indicator	Definition	Sustainability range	Desired development[1]	Unit
Return on investment	Earnings before interest and taxes (EBIT) divided by total capital	2.76 - 20	+	Percent
Labour productivity	Gross value added / labour invested	not used for SSP[2]	+	CHF/h
Hourly wage	Hourly wage in each value added level / average hourly wage of the Swiss population	75 - 125	+	Percent
Market power	Expert assessment of the negotiating power of the levels: producer, processing and consumer	-2 – 2	+	Value between -4 / +4
Social capital	Ratio share stakeholder representation of each level in the parliament related to the share of labour force	1 – 2	-	Value
Social acceptance	Willingness to pay of Swiss population for agriculture, measured as percentage of the population considering the subsidies to agriculture adequate ("in etwa richtig")	50 -100	+	Percent
Human capital (I)	Number of employees per production level	20'000 – 56'599	-	Value
Level of education	Percentage of employees with educational level ""practical experience", basic education" and "higher education"	37 - 100	+	Percent
Subsidies	Subsidies paid to the milk industry as percentage of the gross yields	0-36	-	Percent

1: Defined together with stakeholder and involved experts
2: The indicators were excluded from the sustainability space, based on the system analysis results (Schmid, 2008)

The sustainability solution space

Fig. 4 shows the results of the sustainability solution space. In grey the normalized values of the indicators are presented (corresponding to the sustainability ranges), blue the current values, the black arrows represent the trends and green the sustainability solution space if all the interactions among the indicators are considered. Even if the interaction among the indicators is not considered, the current values of the indicators electricity consumption production, biodiversity Switzerland, social capital, the hourly wage and the return on investments are outside of the sustainability ranges defined. The trend analysis furthermore shows that the indicators electricity consumption production, and social capital are developing in the wrong direction, whereas the hourly wage and return on assets show a trend in the direction of more sustainability.

If the interaction between the indicators is considered it can be observed that the sustainability solution space is smaller than the sustainability ranges defined. In this case also the indicators greenhouse gas emissions, number of employees and educational level are outside of the sustainability solution space.

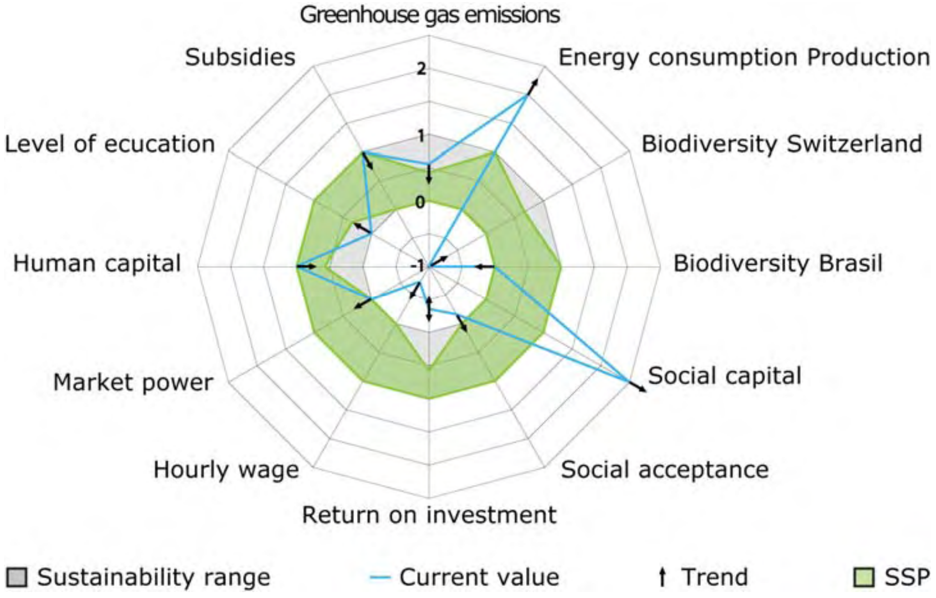

Fig. 4**:** Sustainability solution space short-term scenario (after Schmid, 2008; Binder *et al*., 2008).

Discussion

The geometric SSP model of indicators and their relations allows a more complex understanding of the system and its linkages. In this context, attaining sustainability can conceptualized in two ways. On one hand, attaining sustainability can be done by transforming the system such that it comes to lie within the SSP volume, or maintaining it within the volume boundaries if it is already inside. On the other hand, attaining sustainability can be done by changing the framework conditions: the indicator boundaries or their relations to each other. Changing the framework conditions could be done at the policy, regulation or market levels. The SSP model thus provides a tool to analyse the system framework, and can therefore be used in a multi-stakeholder context as a basis for discussion of potential alternatives.

Further research should study the relationship between the sustainability solution space and the objective space in decision analysis. In addition, potential changes in the setting of the lower and upper boundary due to cultural boundary conditions should be investigated. Still one open question is to which extent interval computation or functional relationships beyond the linear relationship used in the presented approach might improve the results obtained.

Conclusion

This paper developed a sustainability solution space for the milk value added chain of Switzerland. 17 indicators (8 ecological and 9 socio-economic) were used to describe the system. A geometric approach, considering the interaction among the indicators, as well the sustainability ranges, which were elicited in a stakeholder process, allowed for calculating the sustainability solution space. The results show that:

- The consideration of the interaction among the indicators significantly influences the results
- The developed tool can be used to assess the current situation and analyze the effect of strategies and envisioned policies

The tool will be further developed for allowing for an in depth trade-off analysis and real time simulation of strategies.

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Energy use in the life cycle of frozen concentrated orange juice produced in Brazil

L. Coltro*, A.L. Mourad, S.P.M. Germer, T.A. Mendonça and R.M. Kletecke
Institute of Food Technology – ITAL/CETEA, P.O. Box 139, ZIP 13070-178, Campinas, SP, Brazil
*email: ledacolt@ital.sp.gov.br

Keywords: frozen concentrated orange juice (FCOJ), fruit, LCA, Brazil; sustainability

Abstract

Brazil is the world's largest orange producing country, with a total planted area of more than 820,000 hectares. The bulk of the oranges produced in Brazil (70%) is processed into frozen concentrated orange juice (FCOJ) by large processing companies. Exports represent around 97% of the total FCOJ produced, making Brazil the largest world producer and exporter of FCOJ. The Brazilian citrus sector accounts for half of the world's supply of orange juice and for 80% of the juice traded on the international market. In 2003, the citrus production was 13.3 million tons only in the State of São Paulo, the biggest Brazilian producer region (79%). The goal of this paper is to present the energy use at the several steps of the life cycle of FCOJ produced in Brazil. The scope of the whole work was to qualify and quantify the main environmental aspects of FCOJ produced in Brazil in order to establish parameters for the discussion on the good environmental performance of Brazilian FCOJ. The results showed that approx. 70% of the energy use is due to orange cultivation since the proportion is 10:1 of oranges for producing FCOJ.

Introduction

Brazil and United States are the two largest citrus producers, followed by China, according to USDA (2004a). In Brazil, the State of São Paulo accounts for approx. 80 percent of the total citrus production, with an average yield of 20,200 kg/ha (Ministério…, 2008). More than 90 percent of the orange crop is processed into frozen concentrated orange juice.

Exports of orange juice during 2002/03 from the major producing countries were approx. 1.3 million tons (65 degrees brix), as shown in Tab. 1. Exports from Brazil, which accounts for over 80 percent of the total, were over 1 million ton. Brazil's consumption of processed orange juice was approx. at only 18,000 tons during 2003/04 (marketing year July 2004-June 2005), representing only approx. 1 percent of production. Brazilian consumers are more likely to fresh squeeze oranges for their juice needs, rather than purchase orange juice (Ministério…, 2008; USDA, 2004b).

Tab. 1: Major producing countries of citrus and orange juice in 2002/03.*

Country	**Total citrus production**		**Orange juice**				
			Beginning stocks	**Production**		**Exports (65°Brix)**	
	1,000 t	**%**	**t**	**t**	**%**	**t**	**%**
Brazil	**14,974**	**22.0**	**240,000**	**1,005,000**	**46.9**	**1,135,000**	**83.2**
USA	13,768	20.2	473,767	898,289	41.9	73,299	5.4
China	10,145	14.9	0.0	1,800	0.0	3,406	0.2
Spain	5,757	8.4	2,000	56,000	2.6	56,000	4.1
Mexico	5,010	7.3	3,000	13,000	0.6	9,900	0.7
Others	18,575	27.2	33,792	170,414	7.9	86,352	6.3
Total World	68,229	100	752,559	2,142,703	100	1,363,957	100

* USDA, 2004a; USDA, 2004b; Ministério..., 2008.

Despite orange juice is the major orange product, several by-products are produced during the orange processing, as shown in Tab. 2 (ABECITRUS, 2004; Tetra Pak, 1998). These products have several applications in the internal and external market, including production of chemicals and solvents, aromes and fragances, inks, cosmetics, animal feed, etc.

Tab. 2: Products derived from whole oranges (ABECITRUS, 2004; Tetra Pak, 1998).

1,000 kg oranges	553 kg Juice	0.1 kg Essence oil
		1.1 kg Essence aroma
		100 kg 65° Brix concentrate
		452 kg Evaporated water
	30 kg Pulp	
	3 kg Peel oil	
	413 kg Peel, rag and seeds	

Brazil exports showed a growth of more than 60 percent in the exported volume in a period of 12 years. The citrus production showed a growth of 37 percent in the same period, representing a total of more than 13.3 million tons in the 2002/03 crop only for the State of São Paulo, the biggest Brazilian producer region (79 percent) (ABECITRUS, 2004).

The FCOJ production and export sectors comprise 11 processing industries and 29 thousand farms located in the São Paulo State, which employ 400 thousand workers and generate another 3 million indirect jobs (EMBRAPA/ IAC, 2000).

Citriculture is the primary economic activity in 331 cities located in the São Paulo State, in addition to another 15 towns in the neighbouring State of Minas Gerais, and yields 1.5 billion dollars annually. This sector employs the newest technologies and operates excellent transportation and distribution systems (EMBRAPA/ IAC, 2000).

Method

This study was conducted in accordance with the guidelines and requirements set forth in International Standard ISO 14040 (ISO 14040, 2006).

Goal and scope definition

The goal of this paper was to present the aspects of energy use for FCOJ produced in Brazil for two orange-growing regions (Northern and Southern regions of the State of São Paulo). The scope of the whole study was to establish parameters for the sustainability and a future ecolabelling program for the Brazilian FCOJ.

Functional unit

The functional unit adopted in this study was the production of 1,000 kg of FCOJ. This unit is not related to the function of the FCOJ, since the use stage was not included in the system. Thus, the cradle-to-gate LCI basis was adopted.

System description

The system evaluated includes orange-growing on at commercial farms, harvesting, storage, transport by trucks to the processing plants and orange processing to FCOJ and by-products. This study evaluated the agricultural production of Pêra, Valência and Natal oranges in the Northern and Southern regions of the State of São Paulo.

The characterization of the Brazilian orange producers in terms of farm size, cultivated varieties, watering system and tillage practices was published elsewhere (Coltro et al, 2009).

Due to the several by-products of this system (essence oil, aqueous essence, oil essence, d-limonene and animal feed) it was made an expansion of the system in order to exclude the contribution of the animal feed (the main co-product) and to express the results for FCOJ production.

System boundaries

The stages taken into account were the fertilizers production, the production of energy (directly used in the agricultural operations to operate farm machinery and watering systems and also in the transportation steps and orange processing plants), the orange cultivation and the orange processing plants (Fig. 1). All the transport steps have been taken into account, except the importation of some items like the active ingredients of the fertilizers. The production of capital goods (agricultural machinery, watering pumps, buildings, etc.) and labour have not been included.

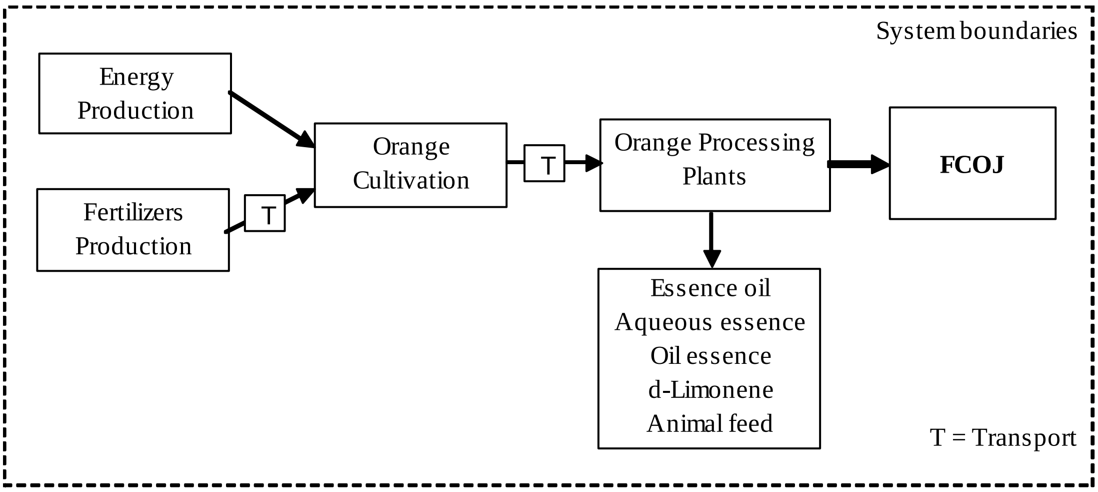

Fig. 1: System boundaries adopted in this study.

Data collection and quality

All information and data considered in this study were taken up in-depth data collection and evaluation by questionnaires either filled in directly on the farm and industry manager or completed by the farm and industry manager and sent in by mail and included the inputs of water, energy, raw materials, fertilizers, pesticides, soil correctors, transport and emissions to water, air and soil.

Farm specific data along with agricultural and industrial production data have been combined in order to construct an FCOJ production model.

Models developed at CETEA for expressing the load transportation and the electric energy production in Brazil in terms of LCA were adopted from Coltro *et al*., 2003 and Madi *et al*., 1999.

The environmental aspects relative to the fertilizers production were taken from recognized database and included in the boundary.

Representativeness and time-frame

The data refer to a production of 367,200 metric tons of orange, 4 million plants in commercial production and an evaluated area that accounts for 19.5% of total orange production of the State of São Paulo. The data reflect the cultivation profile of 30 orange farms and 2 processing industries. Two Brazilian orange producer regions located at the Northern and Southern of the State of São Paulo were evaluated.

Time-related coverage refers to the 2002/2003 crop (marketing year July 2003-June 2004).

Results and Discussion

The citrus production is increasing in the Southern of the State of São Paulo, changing the citrus production map of the State. Besides the enlargement of the planted area in this region, they are employing high technologies. For that reason, two regions for orange production were considered in this study: Northern (traditional orange producer region) and Southern of the State São Paulo. This methodology was defined considering the edafo-climatic characteristics and phytosanitary problems (Guilardi, 2002).

Fig. 2 shows the energy use along the life cycle of the FCOJ produced in Brazil for the functional unit of 1,000 kg. The total energy accounts the all upstream energy use to deliver energy in, for instance electricity (from hydroelectric, fossil fuels and nuclear power plants) and fossil fuels.

Since 50% of the total energy used by the system is from renewable resources, the GWP of this product is related to approx. 70% of the total energy use (taking into account the methane emission from water dams of the hydroelectric power plants). The renewable energy is related to the majority of hydroelectric power plants in the Brazilian energy grid (Coltro *et al.*, 2003).

The results showed that the major energy use is attributed to the orange cultivation (71%), followed by the FCOJ production (23%) with a small contribution of the transport steps (6%).

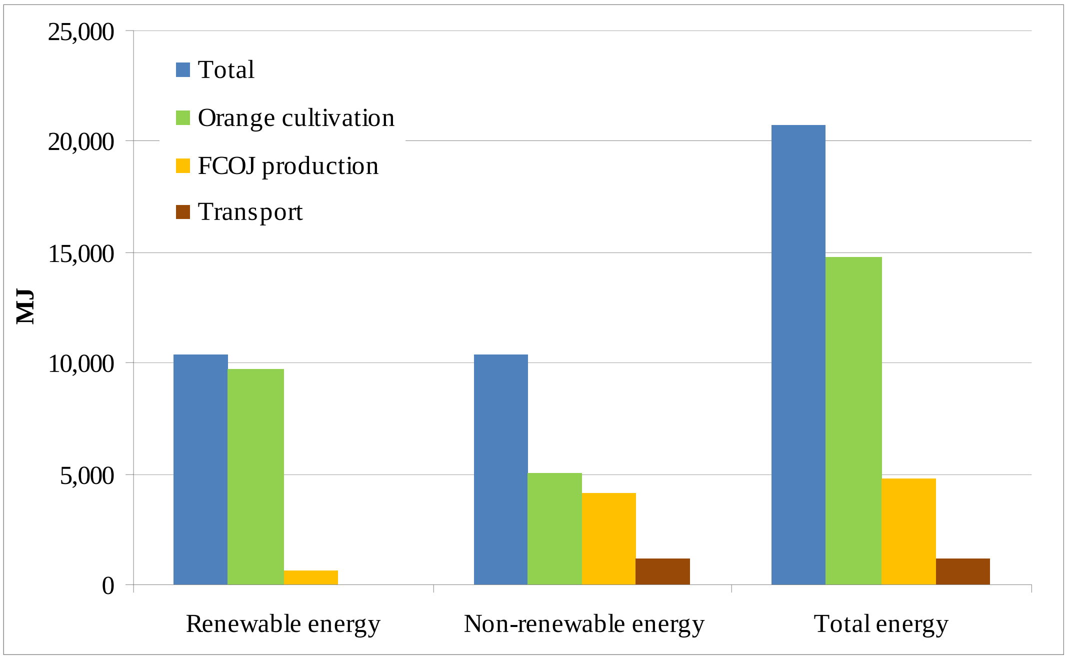

Fig. 2: Contribution analysis of the energy use for producing 1,000 kg FCOJ.

The greater contribution of the orange cultivation is related to the amount of orange used for FCOJ production, i.e. approx. 10 tons of orange are used for producing 1 ton of FCOJ. This large proportion enhances the contribution of this stage in the life cycle of the FCOJ. Nevertheless, approx. 65% of the energy used in this stage is renewable and then, with much lower contribution to the GWP of the product than non-renewable energy resources.

A high energy use was also observed for the agriculture step of the LCA of integrated orange production in the Comunidad Valenciana, Spain (Sanjuán et al, 2005). In this case, the energy was based on fossil fuels and the conclusion is that the energy dependence of agriculture on these energy sources should be reduced by replacing them for alternative renewable sources in order to reduce the environmental impact.

Approx. 86% of the energy use in the FCOJ production stage is non-renewable. The reason for this higher non-renewable energy use is due to the energy consumption for concentrating the orange juice and for keeping it frozen.

Conclusion

This study supplied important results for better understanding the contribution of the agricultural practices and the industrial steps to the potential environmental impacts of the FCOJ production.

Approximately 70% of the energy use is due to orange cultivation since the proportion is 10:1 of oranges for producing FCOJ.

The GWP of FCOJ is related to 70% of the total energy use since it is mainly related to the amount of non-renewable energy used in its life cycle.

Acknowledgement

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Cradle to gate study of two differing Brazilian poultry production systems

V. Prudêncio da Silva Júnior[1], S. R. Soares[1], R. A. F. de Alvarenga[1]
[1] Universidade Federal de Santa Catarina, Centro Tecnológico, Departamento de Engenharia Sanitária e Ambiental, CEP: 88010-970, Florianópolis/SC – Brasil. E-mail: vamilson@ens.ufsc.br

Keywords: Life cycle assessment; poultry; animal production, Brazil; production intensity; production scale

Abstract

Associated with the strong growth of poultry production in Brazil, environmental impacts caused by this activity appear as a problem. An important characteristic of the poultry-production chain is its spatial decentralization (each phase may take place in different locations within one or several countries). Life Cycle Assessment (LCA) has the ability to analyze systems independent of time and space and the potential environmental impacts associated with a specific product (Basset-Mens 2005). However, the diversity of poultry-production systems and the lack of knowledge about the environmental performance of different systems are factors that render difficult the use of this tool for impact evaluation of poultry production. Thus, it is necessary to perform site-specific studies to adapt LCA tools for Brazilian poultry. This work describes two current supply chains of poultry production in Brazil: the southern system, characterized by decentralized production in small farms with feeds obtained from other states (Spies, 2003), and the central-west system, with feeds produced within the farm and located relatively far from industrial areas. The description shows the environmental aspects and important points for the accomplishment of a LCA study, emphasizing the distance of transportation as a factor predominant. Based on a previous LCA study of poultry production in western Santa Catarina, we performed a LCA for the central-west supply chain, adjusting only the distances involved in all stages of transport and comparing the results. Accepting as true the hypothesis that almost all stages in the life cycle are similar in the two supply chains, the comparison results showed lower environmental impacts per tonne of frozen chickens delivered to the port in the central-west of Brazil supply chain.

Introduction

The growth of the poultry industry in Brazil has been highlighted in recent years. Total production of chickens in Brazil increased from 2 million tonnes in 1989 to 9.7 million tonnes in 2007, of which the three southernmost states, *Paraná*, *Santa Catarina* and *Rio Grande do Sul*, are responsible for approximately 54% (ABEF, 2007). As a result of this growth, environmental impacts related to chicken production increased, encouraging this study.

Life Cycle Assessment (LCA) is one of many tools developed for the evaluation of environmental impacts of production systems. It was initially developed for industrial applications but more recently has been used for analysis of agricultural production, especially for single-crop production systems or processes of food production on an industrial scale (Caldeira-Pires *et al.*, 2002). LCA has shown to be a viable tool for analyzing impacts on agricultural systems (van der Werf & Petit, 2002), and therefore it is appropriate to adopt this approach in this research.

While poultry production in southern Brazil has been consolidated and geographically extended, the situation for grain production in the region was affected by the market due to the increasing supply of grain in other regions of the country, mainly the centre. Currently there is a trend of expansion of poultry farming to the central west and north, mainly *Goias*, *Bahia*, *Minas Gerais* and *Mato Grosso* states. The main reason for this expansion in activity in the central west is the proximity to areas supplying raw materials at low cost, mainly corn and soybeans for feed (Faveret Filho & Lima de Paula, 1998). The need to increase the quantity of integrated farmers also contributes to the production process. The new regions are characterized by the dominance of producers with larger scales of production and bigger farms than those of southern Brazil. Companies like *Sadia* and *Perdigão* already have established plants in the central west, looking for strategic and logistical advantages, keeping the model of vertical integration and getting advantages in the economy of scale (Faveret Filho & Lima de Paula, 1998).

Currently, problems related to animal welfare has been concern in various segments of society, especially in intensive production systems. However, studies using the LCA approach normally do not consider issues related to animal welfare, due to the difficulty and subjectivity of obtaining animal welfare indicators. In this study, which compares two Brazilian systems of poultry production, the aspect that more is come close to a welfare indicator is the amount of birds per m². Meantime, aspects of animal welfare are not considered in this study.

To quantify the environmental impacts (local, regional and global) of these two supply chains, we intended to compare them with LCA; however, preliminary information suggests that certain steps can modify the results significantly if they are (or not) included in the scope of LCA. Among these, the various distances involved in the stage of transport is the factor that most likely affect the results of LCA. This study aims to determine what are the environmental impacts in both regions studied, whereas, initially, only the distances are different between the two systems.

Approach

First, two representative supply chains of poultry production in Brazil were defined and characterized:

- Standard family-based industrial chicken (traditional) – western *Santa Catarina* in **S**outhern **B**razil (**SB**)
- Standard industrial chicken (recent) – **C**entral-**W**est of **B**razil (**CWB)**

Stages with greater influence on the environmental performance of chicken production from a LCA viewpoint were highlighted. Both supply chains were characterized based on bibliographic data of studies already conducted in Brazil. Then, based on a previous LCA study of poultry production in western *Santa Catarina* (SB), the stages with greater contribution to environmental impacts were identified and compared with the same stages in the central-west region (CWB).

As for the moment, we do not have data for other stages of the life cycle in both supply chains, this work is restricted to run the same LCA conducted for the South of Brazil, adjusting the distances involved in all stages of transport, according the situation in Central-West, and comparing the results. However, the previous study did not consider the stages of slaughtering, processing, freezing and transport to the port. These steps were considered in this study despite in simplified way.

Although the characterization method "Eco-Indicator 95" is superseded, it was used in the previous LCA study (Spies, 2003) and we chose to keep it to compare the results, thus using the same impact categories and the same criteria.

Characterization of production systems

Taking the current situation of poultry farming in Brazil into account, these two different supply chains of chicken production are representative of the national situation (Fig. 1). The pioneering system, already well established in the south (SB), is characterized by the predominance of small properties located relatively far from grain-producing regions. The more recent system (CWB), located in the Central West, is characterized by the predominance of large properties within the grain-producing region.

In both cases, during the industrialization process large amounts of liquid and solid wastes are generated that require appropriate separation and treatment before being released into the environment. Because consumption patterns have changed (chicken cuts as opposed to whole chickens), the generation of by-products has increased in recent years (Fernandes, 2004). Several of these by-products and wastes have economic value and can be used after processing.

The stages of chicken production, slaughter and processing are similar in the two chains and may not have significantly different environmental impacts. Therefore, this study considers this stages in a simplified way. The dotted rectangle in Fig. 1 represents the scope of the LCA which had been held for the region SB.

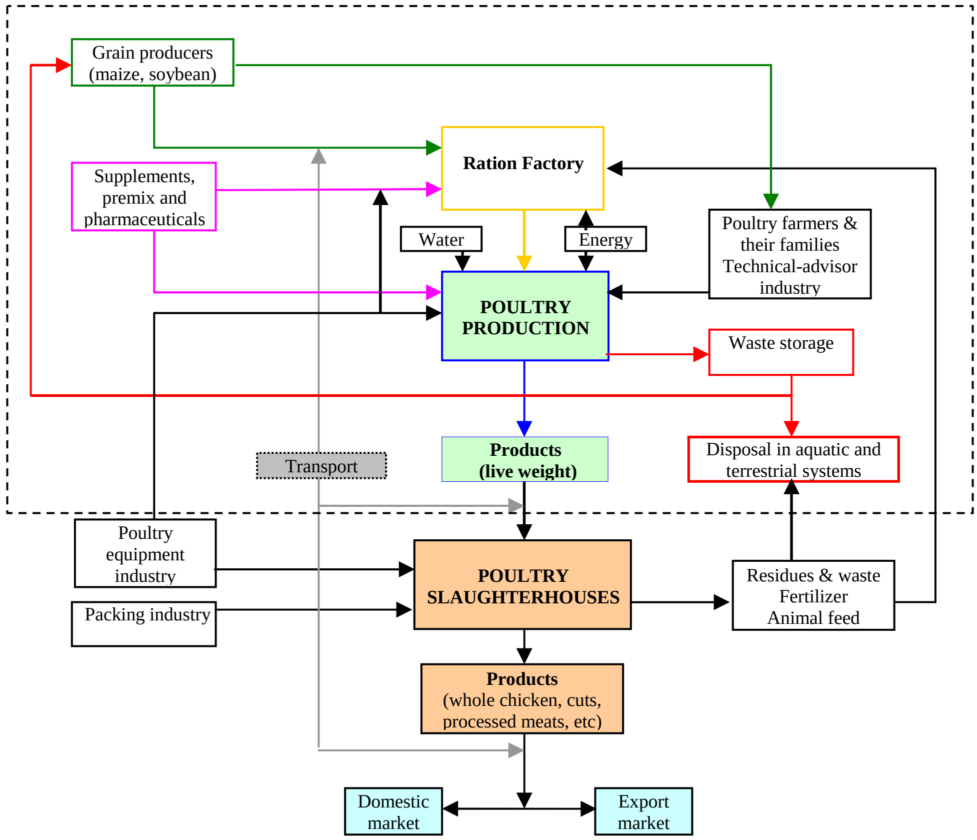

Fig. 1: Simplified flow of a poultry production system, adapted from Spies (2003).

LCA in SB supply chain

In the LCA made for SB, according Spies (2003), the approach used was “Streamlined Life Cycle Assessment”- SLCA. In that study, a SLCA of pig and poultry production at farm level was conducted to identify and quantify the factors that have the greatest environmental impact (hot-spots). The study compared the environmental burdens of each production system, and tested the hypothesis, derived from empirical data provided by an e-survey of stakeholders, that poultry production has lower environmental impact than pig production. The stages considered were production of maize and soybeans; transportation of soybeans to crushers; extraction of oil and production of soymeal; transportation of maize and soymeal to feed factories; production of poultry feed; transportation of feed to poultry producers; production of chicken on the farm; disposal of waste (litter) from chicken production; and transportation of chickens to slaughterhouses (Spies, 2003). The functional unit (FU) adopted was the production of 1 tonne of live-weight chicken delivered to the slaughterhouse gate. The categories of impact considered were climate change; ozone depletion; acidification; eutrophication; heavy metals; carcinogens; pesticides; energy resources; solid-waste emission (Method: Eco-indicator 95, Europe). See more details of the approach used in Spies (2003), chapter 7.

The LCA made for SB showed that the stages involving production and transportation of feed (including production and transportation of grain) have an important effect on impact categories. In terms of climate change, the results showed that feed production caused about 63% of the impact of poultry production, of which 94% was due to CO_2 emissions, 5% from N_2O emissions, and 1% from CH_4 emissions (Spies, 2003). For ozone depletion, feed production caused 89% of the impact, all due to Halon-1301, released mainly during transport. For acidification, feed production caused 49% of the impact, of which 84% was caused by the release of NH_4. For eutrophication, feed production caused 60% of impact, of which 54% was caused by NH_4 emissions.

Differences between the two supply chains

The flowchart shown in Fig. 1 is common to both systems; however, to establish the main differences between them, the work of França *et al*. (2007) was consulted. They surveyed 42 properties in the CWB supply chain (*Rio Verde, Goiás* state) and 104 properties in the SB supply chain (*Videira, Santa Catarina* state).

They observed that 86% of the farms in the CWB supply chain were medium (30-200 ha) and large (>200 ha), compared to 14% in the SB supply chain. The level of technology also appeared an important difference. While SB systems had both manual and automated poultry production, all CWB systems were automated. Moreover, only 1% of chicken houses in SB had air conditioning, while all buildings in CWB had air conditioning (Franca *et al*, 2007). This technological standardization in the CWB chain increases production efficiency and significantly improves farm management for integrators. This situation contrasts with SB, where several combinations of technologies are used. Tab. 1 shows the technical performance indicators for the both systems.

Tab. 1: Performance indicators for southern Brazil (SB) and central-west of Brazil (CWB) supply chains.

Indicator	SB	CWB
Breeding age (days)	42	42
Final weight (kg live weight – LW)	2,48	2,4
Birds per m²	11,7	15
Mortality (%)	4,39	4,16
Daily weight gain (g LW/day)	57,62	55,71
Feed conversion (kg feed/kg LW)	1,86	1,89
Batches per year	6,4	6

The feed conversion is a major aspect for the LCA, since most of the impact is associated with production of the feed. Therefore, to produce a certain amount of chicken, if more feed is required, greater the environmental impact.

The carrying capacity of trucks used in each system also differs. For transporting feed, the loading capacity of a truck is 13 tonnes in SB and 26 tonnes in CWB (Franca *et al*, 2007). Consequently, the transport capacity jumps from 3131 to 7178 birds/truck, respectively. Compared to SB, this reduces logistics needs by 22 trucks for feed transport and 23 trucks for broiler transportation in CWB, as well as reducing the amount spent on wages for their drivers (this calculation based on the total number of producers interviewed for the same amount of chickens produced in both regions). Another logistics improvement in the CWB supply chain compared to that of SB is the number of integrated farms; therefore, for the same amount of product there is a need for more integrated farms in SB in CWB.

In the SB supply chain, the mean distance between the grain (maize and soybeans) producing regions and grain storage is 250 km; from grain storage to the feed factory the mean distance is 600 km, which gives a total distance of 850 km (Spies, 2003). The mean distance between poultry farms and slaughterhouses is 95 km, while that between the feed factory and farms is around 42 km (Martins *et al.*, 2007). The distance of incubators from poultry farms is around 100 km, and the port is located around 544 km from the slaughterhouses (Martins *et al.*, 2007).

In the CWB supply chain, the maximum distance between the farm and slaughterhouses is 60 km. The maximum distance between grain producing regions and the feed factory is 120 km (Faveret Filho & Lima de Paula, 1998). The mean distance between the feed factory and incubators in CWB has not yet been determined, though it is less than or equal to that in SB. The establishment of these mean distances affects the LCA results, especially the emission of CO_2 equivalents by transportation.

The greatest contrast is the distance between slaughterhouses and the port for international export: 1650 km in CWB (Carfantan, 2007) and 544 km in SB (Martins *et al.*, 2007).

Based on the bibliographic data characterizing both supply chains, the following table summarizes the main differences found:

Tab. 2: Main differences between southern Brazil (SB) and central-west of Brazil (CWB) supply chains.

Main differences	SB	CWB
Grain production region to feed factory	850 km	120 km
Feed factory to poultry farm	42 km	< 42 km*
Poultry farm to slaughterhouse	95 km	60 km
Incubator to poultry farm	100 km	< 100 km*
Slaughterhouse to port	544 km	1650 km
Size of sheds	1200 m² x 1	1600 m² x 4
Technology level	heterogeneous	high and homogeneous
Worker : bird ratio	1 : 9713	1 : 34,885
Capacity of feed truck	13 t/truck	26 t/truck
Capacity of chicken truck	3131 birds/truck	7178 birds/truck
Number of integrated farmers	larger	smaller

* = information not found in literature, presumed by the authors.

Results

The results of LCA for the two supply chains can be summarized in the Tab. 3.

Tab. 3: Impacts characterization to produce 1 tonne of frozen chickens delivered to the port of Itajaí (SC, Brazil), for southern Brazil (SB) and central-west of Brazil (CWB) supply chains.

Impact categories	Units	SB	CWB
Greenhouse	Kg CO_2	2,583	2,250
Ozone layer	Kg CFC_{11}	0	0.000405
Acidification	Kg SO_2	94.9	89.9
Eutrophication	Kg PO_4	25.4	24.62
Heavy metals	Kg Pb	0.01	0.0103
Carcinogens	Kg B(a)P	0	1.39E-5
Pesticides	Kg active subst.	0.78	0.77
Energy resources	MJ (low voltage)	24,850	20,483
Solid waste	Kg	26.01	25.86

Analyzing the results of the comparison by stages (Fig. 2), the stages of feed production and transport of live chickens are more favourable to the CWB system in almost all impact categories. However, the transport of frozen chickens to the seaport results favourable to the SB system in all impact categories. Analyzing the overall results (all stages) the CWB is better for the categories of greenhouse and energy resources. Fig. 2 shows these differences.

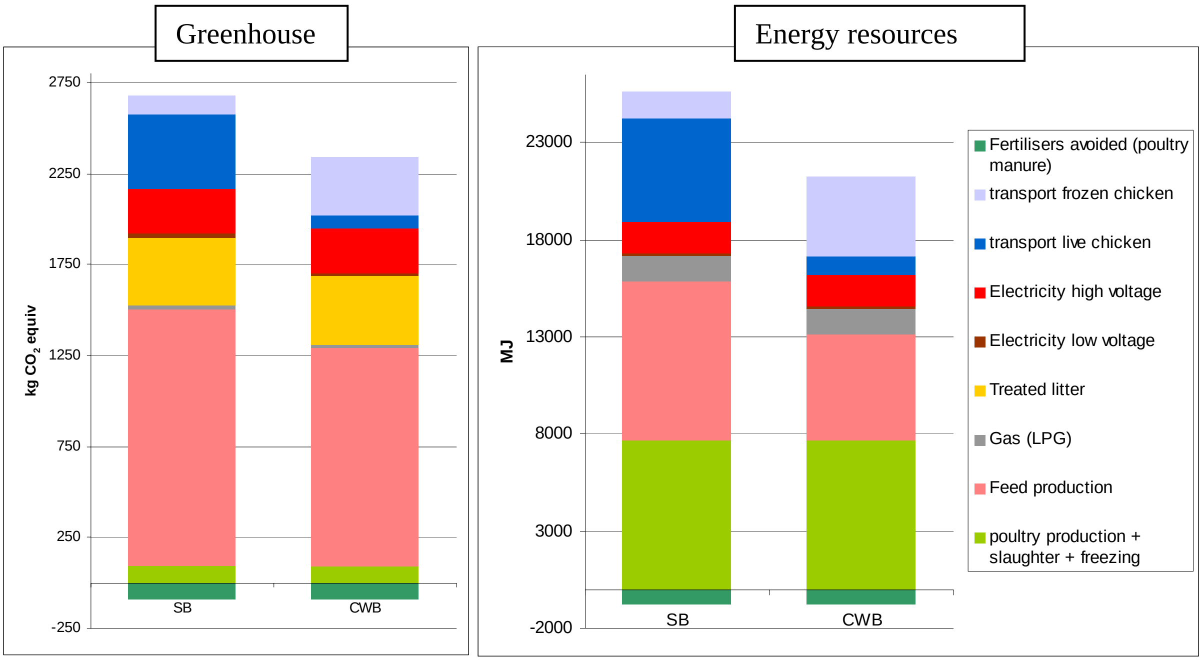

Fig. 2: Environmental impacts of 1 tonne of frozen poultry delivered to the port for SB and CWB systems by *greenhouse* and *energy resources* impact categories and life cycle stages.

Discussion

To perform a LCA for the SB supply chain, several scenarios had to be chosen, although in practice the scenarios act together. For example, maize and soybeans have several possible production systems, such as conventional tillage, zero tillage, minimum tillage, production with or without irrigation, and other variations. Certainly impacts vary in each case; however, a description of the management practices used has been chosen to represent this step. In the LCA for SB, the impacts related to grain production are embedded in the feed production. Uncertainties like these need to be organized hierarchically, detecting those that contribute most to the impacts; this can be done with a sensitivity analysis.

According to Spies (2003), the LCA method (Eco-indicator-95) used to characterize the impacts of the SB system is based on hypothetical modelling methods that do not agree fully with the reality of *Santa Catarina*. Since many hypotheses are based on European conditions, there is a degree of uncertainty about assumptions at the regional level. The main differences probably involve grain production and transport and assimilation of waste due to different climatic conditions in Brazil.

Keeping in mind the future implementation of LCA for both supply chains, it is necessary to identify which stages have particular interest from an environmental viewpoint, despite the uncertainties. Due to the burning of fossil fuels for transport, distances to be travelled along the chain and truck capacity are crucial in determining environmental impacts. Another important issue is the size of farms, according to economy of scale, which affects the logistics of transport, energy consumption, food, water, and use of equipment involved.

The work already done on the SB supply chain shows that in all impact categories the largest contribution came from feed production, which includes the production of maize and soybeans and their transport. Depletion of energy resources is caused mainly by transport, heating, and electricity used in poultry production. The use of treated poultry litter as fertilizer reduces the balance of impacts significantly, particularly local impacts, such as heavy metals, eutrophication, and acidification.

Assuming similarities in both supply chains for maize and soybean production, extraction of soybean oil and production of soymeal, feed and poultry production, and manure disposal, the differences between the chains become more evident with respect to the distances involved between these stages. This study shows that a LCA of the CWB supply chain using parameters similar to those used in SB will predict less severe impacts per tonne of live weight. Even if differences exist in other stages, such as fully-automated vs. manual poultry production, the largest impact contributor would remain feed production and transportation because it has the most influence among all impact categories considered.

However, the LCA of the SB supply chain (Spies, 2003) had system limits that stopped with the delivery of live chickens to the slaughterhouse. In this study the functional unit adopted was different (frozen chicken delivered to the port), including some new stages, and the potential impacts was quite different between the two chains given the distance between each and the main port used as a route for export of chicken in Brazil (*Itajaí*, *Santa Catarina*).

Conclusion

To improve evaluation of a system with LCA, researchers can consider organizing uncertainties hierarchically according to the purpose of the analysis, adjusting the study’s scope, and performing sensitivity analyses. The variability shown suggests that it is not likely to obtain a representative dataset of chicken production system for the whole country, without incurring any errors. Most of this variability is in the production of grains for feed. Due to the large size of the country, many cultures may have different models of production, such as “no tillage” or conventional systems, or use/non-use of organic fertilizers, etc. These variations are more evident for maize, which is the main component of animal feed. However, likely for other products such as sugar and ethanol, there is less variation, as the practices are more homogeneous for the cultivation of sugar cane.

Accepting as true the hypothesis that maize and soybean production, soybean-oil extraction, feed and poultry production, and manure disposal are similar in the two chains, an LCA comparison results in predictions of lower environmental impacts per tonne of frozen chickens delivered to the port in the CWB chain.

The scope of LCA to be defined for the two supply chains will fundamentally influence the results. Nevertheless, in this study, both the production of a tonne of live chicken delivered to the slaughterhouse gate, or a tonne of frozen chicken delivered at the port, had lower level of environmental impact to the CWB supply chain.

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Environmental assessment of Filipino fish/prawn polyculture using Life Cycle Assessment

Baruthio A.[1,2,4], Aubin J. [1,2], Mungkung R.[3], Lazard J. [4], Van der Werf H.M. [1,2]
[1] Institut National de la Recherche Agronomique (INRA), UMR 1069, Soil Agro and hydroSystem, F-35000 Rennes, France, aurelle.baruthio@etu.ensat.fr

[2] Agrocampus Rennes, UMR 1069, Soil Agro and hydroSystem, F-35000 Rennes, France

[3] Department of Environmental Science, Faculty of Science, Kasetsart University, P.O. Box 1072 Kasetsart, Chatuchak, Bangkok 10903, Thailand
[4] Centre de coopération Internationale en Recherche Agronomique pour le Développement (CIRAD), UPR « Aquaculture et gestion des ressources aquatiques », Département Persyst, TA B-20/01, Avenue Agropolis, 34398 Montpellier cedex 5, France

Keywords: extensive polyculture, prawn, environmental assessment

Abstract

The environmental impacts of a representative Filipino polyculture system were assessed using the Life Cycle Assessment method. The former associates the production of four species: prawn, tilapia, milkfish, and crabs. It is considered as an extensive system that is based mostly on natural inputs. The polyculture system was analysed from two perspectives, first by taking into account all four products, and then by applying an allocation that focuses on prawns production only. Results were compared to an intensive monoculture fish-production system (European sea-bass in Greece). The analysis showed that potential acidification and eutrophication impacts per tonne of all polyculture products combined were 36 and 18% higher, respectively, than those per tonne of sea-bass from monoculture, while energy use was 15% lower. When economic allocation was applied to evaluate prawn production only, impacts per tonne of prawn from polyculture were 33-46% higher than those per tonne of all polyculture products.

Introduction

In Asia, aquatic products are the main source of animal protein for the local population, and aquaculture is an ancestral activity. Consequently, aquaculture has to meet several objectives: producing food for an increasing population, providing a source of income, and managing land and water while respecting the environment. The present article focuses on the brackishwater polyculture system of the province of Pampanga on Luzon Island, the Philippines. It is located in an estuary opening onto Manila Bay.

Polyculture in Pampanga has existed for more than 300 years. It developed on mangrove swamps and expanded until the 1970s (Primavera, 1995). Mangrove destruction now is irreversible in that area, and pond conversion for mangrove rehabilitation is no longer an option. Today the polyculture system occupies more than 16,000 hectares of ponds in the province. Three or four species are associated in this system: tiger prawn *(Penaeus monodon)*, crab *(Scylla serrata or Scylla olivacea)*, milkfish *(Chanos chanos)* and in areas far from the sea, tilapia *(Oerochromis niloticus)*. The production is mainly organised around prawn production, which is the most valuable product. The polyculture system has evolved from traditional system; it takes its feed from the environment and uses low stocking densities. Therefore, it is considered as an extensive aquaculture pond system.

Our study proposes (1) estimated the environmental impacts of the polyculture system using Life Cycle Assessment (LCA) and (2) compared them to those of a European sea-bass (*Dicentrarchus labrax*) monoculture system.

Method/Approach

Polyculture system definition

The system boundary includes (1) the hatchery stage, (2) rearing (i.e., "growing-out"), and (3) harvest. Sale, processing, transport and distribution were not included in the system. The analysis takes into account one year of production.

Hatchery stage: Tilapia fingerlings are produced locally in Pampanga Province in freshwater ponds. Milkfish fingerlings either were collected from the sea (for the most grow-out operations) or purchased from Indonesian or Taiwanese hatcheries. Fry-sized crabs were caught by hand in mangrove swamps using a landing net. Prawn broodstocks were caught in the sea using small trawlers. The feeding of larvae was based exclusively on skeletonema algae or balanced with concentrated feeds or brine shrimp (*Artemia* spp.).

Farm stage: Fifteen farms were assessed. Milkfish and tilapia are fed with phytoplankton growing naturally in the pond. Since crabs are stocked at low densities (250-500 per hectare) only prawns are fed, with snails (Horn shell (*Cerithium tenellum*) and Rodong shell (*Telescopium telescopium*)) gathered from the surrounding environment; this operation required pulling a net with a motorboat for 3-7 hours (depending on snail density). Prawn stocking density was 50,000 post-larval prawns per ha. Chemicals and fertilizers commonly used were lime and, less often, urea (16-0-0). Eutrophying emissions were calculated according to a mass balance: nitrogen and phosphorus emissions were assessed according to the difference in mass between the harvested products and the inputs (larvae/fingerlings, fertilizers and feeds). The Theoretical Oxygen Demand of solids emitted was added in the calculation according to Paptryphon *et al.* (2004).

Harvest stage: Three months after being stocked, ponds were drained using pumps.

The impact assessment method used was CML 2 Baseline 2000 (version 2.03 ; Guinée, 2002). The impact categories considered are Climate Change Potential (CC), Acidification Potential (AP), Eutrophication Potential (EP), Non Renewable Energy Use (NREU) and Land Occupation (LandOc). For all LCA analyses, the functional unit considered was 1 tonne of fresh aquaculture products. Environmental impact assessment was calculated using the Simapro® 7.0 software and its databases, original data collected from the field and data provided by previous LCA studies (Aubin *et al.*, 2006)

Comparison of prawn from polyculture with fish from monoculture

We compared potential impacts of this production system to those calculated with the same impact-assessment method by Aubin *et al.* (2009) for a sea-bass marine-cage system in Greece, an example of an intensive monoculture system using electricity produced mainly from fossil fuels as in the polyculture system. The sea-bass farm was located on the Evoikos Gulf, north of Athens, Greece, and was dedicated to growing sea-bass from 2 to 350 g in approximately 16 months. The farm consisted of 12 circular-net cages, each 1100 m^3 in volume, arrayed around a platform used for equipment handling. The depth of water under the cages was 25 m, and the average water current was 3 cm/s. The farm was equipped with boats and land-based facilities for feed and material stocking and net cleaning. Annual biomass production was 256 tonnes. Fish were nourished using dry pellets with an average composition of 45% protein, 12% lipids, and 1.3% phosphorus. The economic feed-conversion ratio equalled 1.8 (Aubin *et al.*, 2009).

Results

Impact assessment of all products of the polyculture system

For this analysis, the Functional Unit was 1ton of products (prawn, crab, milkfish and tilapia).

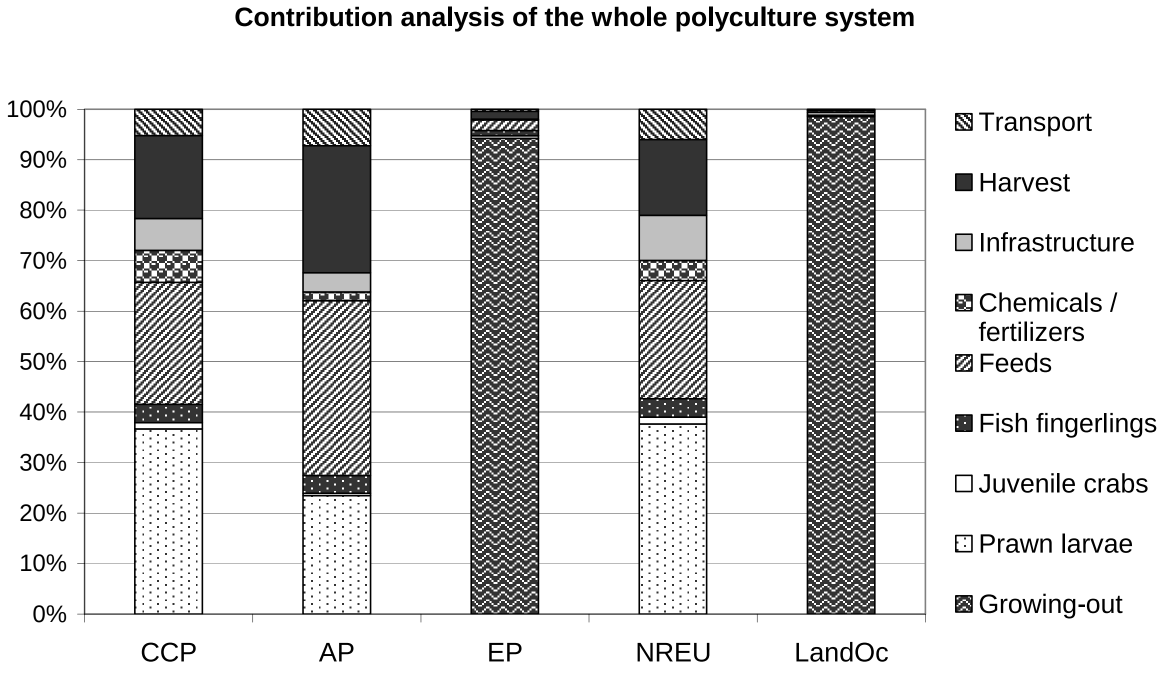

Fig. 1: Contribution of polyculture system components to Climate Change Potential (CCP), Acidification Potential (AP), Eutrophication Potential (EP), Non-Renewable Energy Use (NREU), and Land Occupation (LandOc) impacts for 1 tonne of all products combined. For “Prawn larvae”, “Fish fingerlings” and “juvenile crabs” system components, all stages preceding larvae or fingerling production up to transport to the farm were included.

The major contributor to EP and LandOc was Growing-out (94 and 98% respectively; Fig. 1). Together, Prawn larvae, Harvest and Feeds constituted more than 75% of CCP, AP and NREU impacts.

Comparison of prawn from polyculture with fish from monoculture

Tab. 1 compares potential impacts per tonne of (1) all polyculture products (calculated above), (2) prawns from polyculture, and (3) sea-bass from monoculture.

Tab. 1: Potential impacts per tonne of all polyculture products, prawn from polyculture, and seabass from monoculture. Impacts include Climate Change Potential (CCP), Acidification Potential (AP), Eutrophication Potential (EP), and Non Renewable Energy Use (NREU). The functional Unit was 1 tonne of products.

	Unit	All products (polyculture)	Prawns (polyculture)	Sea-bass monoculture
CCP	kg CO_2 eq.	3553	5108	3601
AP	kg SO_2 eq.	34	48	25
EP	kg PO_4 eq.	129	172	109
NREU	GJ	46	67	55

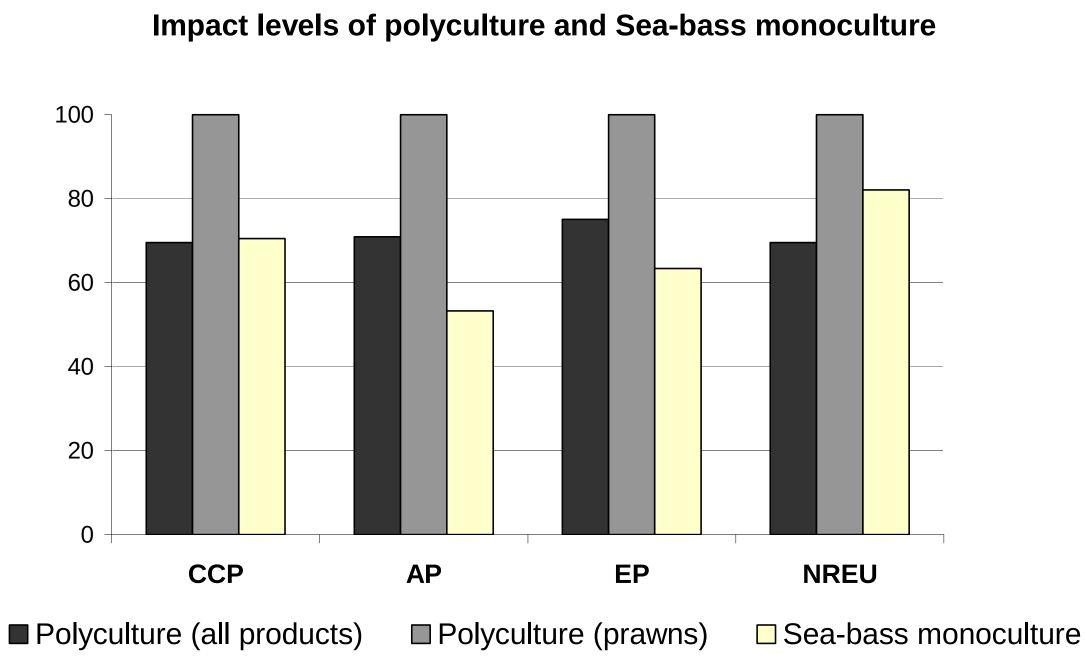

Fig. 2: Relative impacts per tonne of all polyculture products, prawns from polyculture, and sea-bass from monoculture. Impacts include Climate Change Potential (CCP), Acidification Potential (AP), Eutrophication Potential (EP), and Non Renewable Energy Use (NREU).

Since prawn was the most valuable of the polyculture products, its impacts per tonne were 33-46% higher than those per tonne of all products from the polyculture combined (Fig. 2).

When comparing impacts per tonne of sea-bass from monoculture with those of all products from polyculture,

CCP did not differ; AP and EP were 36 and 18% higher, respectively, for all polyculture products than for sea-bass from monoculture, while NREU was 15% lower.

Discussion

For impacts per tonne of all polyculture products, the large contribution of Growing-out to LandOc and EP impacts was due to the low use of crop-based inputs. The large contribution of Prawn larvae was due to their low survival rate, averaging only 5% (Baruthio, 2006), while the large contribution of Harvest resulted from the use of 15-45 l of diesel fuel per ha for pond drainage. The large contribution of Feeds can be explained by the equally large quantities of snails provided (400-5000 kg/ha/cycle), requiring gasoline for collection.

Economic allocation resulted in impacts per tonne of prawn 33-46% higher than those for all products from the polyculture system, since prawn is the most valuable of the four species. Further investigations should be conducted to examine other approaches in impacts allocation for polyculture systems, such as allocation according to physical causality or system expansion, as recommended by Ayer *et al*. (2007).

The higher potential Acidification and Eutrophication impacts per tonne of all polyculture products than per tonne of sea-bass from monoculture reveal the former system’s poor productivity and efficiency, especially due to high prawn mortality and its associated nutrient release. In contrast, the higher NREU per tonne of sea-bass is explained by the energy required for feed processing and growing crop-based ingredients, the main source of greenhouse-gas emissions.

The results emphasise that prawn hatcheries contribute greatly to potential environmental impacts in the polyculture production system due to the low larval survival rate. In response, a few Filipino farmers have chosen to replace tiger prawn with white shrimp (*Penaeus vannamei*), which is not affected by diseases affecting tiger prawn, in particular white spot disease (*Ichthyophthirius multifilis*). This replacement also occurs in Thailand, where most prawn monocultures have converted to white-shrimp production; however, it probably is not a viable alternative for Filipino polyculture systems because they would have difficulty competing with other countries (especially Thailand) that produce white shrimp at a larger scale.

Feeds also contribute greatly to the potential impacts of polyculture. Besides impacts stemming from high fuel consumption (due to collecting snails with motorboats), an apparent decline in snail density has become a concern for local governments. Although alternatives to the use of snails should be sought, the use of concentrated feeds for prawn is likely inappropriate in polyculture systems because of its high cost and the competition for feed with other species. The failure of prawn monocultures in Pampanga in the 1980s and recent experiences in semi-intensive prawn cultures show the maladjustment of such systems to the local environment. Other alternatives need to be considered such as the use of so-called “trash fish”, which are the remaining fish from harvests, as the main source of (natural) feed.

Conclusion

This study aimed to assess the environmental performance of the extensive aquatic polyculture system that constitutes the majority of Filipino aquaculture production. Compared to an intensive fish monoculture using high levels of inputs, it reveals that polyculture performs better in terms of Non-Renewable Energy Use (due to feed origin), but not in terms of Acidification or Eutrophication potentials, due to the low conversion of locally-collected feed (snails) to product biomass in the system.

Aquaculture in Asia mostly converted from extensive polyculture systems to intensive monoculture systems during the 1980s. Knowing that Filipino polyculture derived from a traditional system existing for centuries, that most of its inputs still come from the local environment, and that it is the main source of income (more than 80% of the population depends on it), it is therefore important to maintain and develop it., it is therefore important to maintain and develop it. In addition, considering the worldwide challenge of doubling aquaculture production by 2030 (Kourous, 2007), Filipino polyculture may be a key case study of intensification using the natural productivity of ponds.

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Life cycle assessment of wheat grown in Washington State

R.C. Schenck[1] M. Ostrom[2], D. Granatstein[2], K. Painter[2] and C. Kruger[2]
1 American Center for Life Cycle Assessment, Institute for Environmental Research and Education. PO Box 2449 Vashon, WA 98070. rita@iere.org

2 Washington State University Center for Sustaining Agriculture and Natural Resources. 1100 N. Western Ave. Wenatchee WA 98801

Keywords: Wheat Tillage, Soil carbon, Nitrogen cycle, Herbicide toxicity, Wheat Life Cycle Assessment

Abstract

Over the past several hundred years, the production and international trade of wheat has emerged as a central feature in the development of the modern world diet and agrifood system, with historic environmental, economic and social consequences (Friedmann 1994). The USA is the largest exporter of wheat in the world, exporting 28.5 million tons in 2007. Washington State grows 3.5 million tons of wheat annually, around 90 percent of which is exported overseas, and thus represents about 12 percent of USA wheat exports. This is equivalent to the entire export of wheat from Russia.

The vast majority of the wheat is grown using conventional methods, but growers in Washington are experimenting with different kinds of wheat production to improve farm economic and environmental performance: reduced tillage is a new method currently being tested by different growers. These different approaches imply different impacts. No-till or direct seeding methods typically create almost five times more herbicide ecotoxicity for weed control, but lower fossil fuel consumption.

Searchinger *et al*.(2008) recently raised the issue of land-use change induced carbon emissions in the context of biofuel production. This issue has been the subject of a long-term research program on climate-friendly farming in Washington State. We present here the long-term data from surveys and models on the soil carbon and combined with conventional LCA emissions analysis in the wheat fields of Washington State to provide a weighted average carbon footprint of wheat production, as well as the analysis of wheat produced via different production methods.

Our results show that the greenhouse gas emissions are dominated by the nitrogen biogeochemistry, and that the carbon sequestered in soils are small relative to the emissions of nitrous oxide.

It is intended that this data be used to support the development of climate-friendly farming programs in Washington State, via outreach to growers and consumers.

Introduction

Washington State, located in the Pacific Northwest of the United States, produces approximately three million metric tons of winter wheat for export per year. In Washington State the gradient of rainfall is the primary driver of wheat yields: rainfall in the wheat-growing regions varies from less than 13 to more than 48 cm rain per year. The lower rainfall areas produce dryland wheat in alternation with a fallow year. Wheat is also produced under irrigation, at 56 cm/year. The irrigation comes from dammed rivers via gravity or by pumping from deep wells. The vast majority of wheat is produced using conventional methods. However, there is a growing interest in producing wheat through direct seeded (no-till) methods.

No-till methods employ less fuel and soil disruption, but they use more herbicides to manage the fields. In this paper we examine the interactions among tillage approaches in terms of water and land use, herbicide use and production methods.

Methods and Data

Agricultural practices were based on primary data collected by Washington State University for Sustaining Agriculture and Natural Resources. This data was supplemented by data from the United States Department of Agriculture national Agricultural Statistics Service data for the Washington Wheat Crop in 2006 USDA (2007). Electricity inventory was based on primary fuel distribution data from the local power company and fertilizer production data was based on primary North American data. Where North American Data was not available, comparable unit processes were taken from the Ecoinvent database. Emissions of soil nitrous oxide emissions were based on IPCC guidance IPCC (2003).

The crop rotation in the state is very variable, taking as long as four years between wheat crops and with many different intercalary crops. Sometimes the wheat is the primary crop, and other times it is used solely for the purpose of pest management for a higher-value crop. Impacts were allocated to wheat in the year in which wheat was harvested, except in the case where winter wheat alternated with a fallow year. In that case all inputs for both years were allocated to the winter wheat production. Where the total crop averages are shown, they represent the weighted average of the total crop in the year 2007.

The analysis represents the on-farm production of wheat. Specifically excluded are the production and disposal of infrastructure, buildings and equipment.

Impact assessment was performed using the US EPA TRACI 3.01 methodology Bare *et al.*, (2003), except that eco-toxicity was calculated using the LCA-tox methodology Schenck (2007), a variant of USE-tox.

Results

As noted above and in Fig. 1, rainfall is the primary determinant of wheat yields.

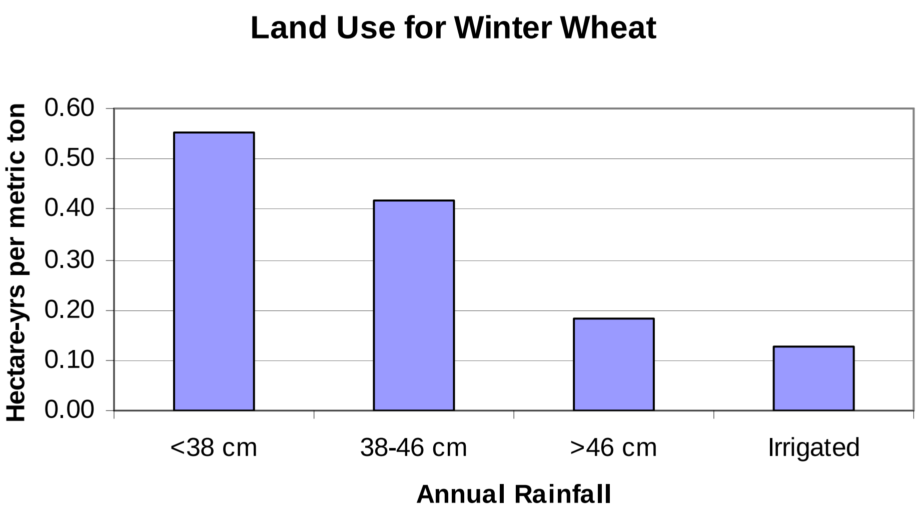

Fig. 1: Land Use for Winter Wheat

Most of the irrigation water for wheat comes from the Columbia River irrigation projects, and flows by gravity to the fields. About one-third of the irrigation water comes from deep wells, which are pumped using electric pumps. Although a relatively small proportion of the fields are irrigated, their high yields dominate the crop production.

Tillage conditions appear to have very little effect on the carbon footprint of wheat production, as can be seen in Fig. 2. The contribution analysis shown in Fig. 3 explains why: the climate change impacts are dominated by the emissions of nitrous oxide from soils. Nitrous oxide is produced in all soils (native or cultivated) when they have fixed nitrogen content and some anaerobic or microaerophillic conditions, as occurs after rainstorms. When energy inputs are reduced as in no-till systems, the dominance of the nitrogen cycle becomes even clearer. Fig. 4 shows that over 80% of the climate change impact is derived from the production and use of nitrogen fertilizers in the no-till system (over 46 cm of rain per year).

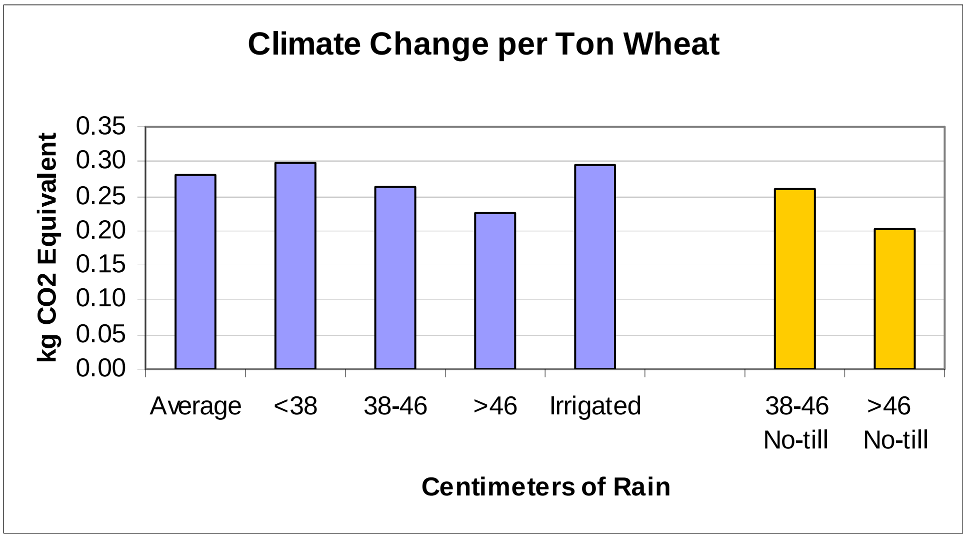

Fig. 2: Climate change per ton wheat

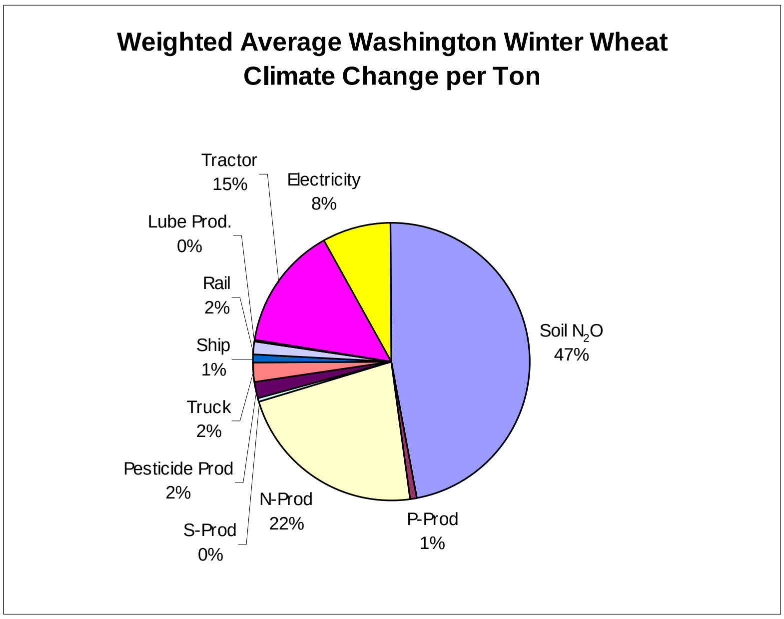

Fig. 3: Climate change contribution analysis weighted average

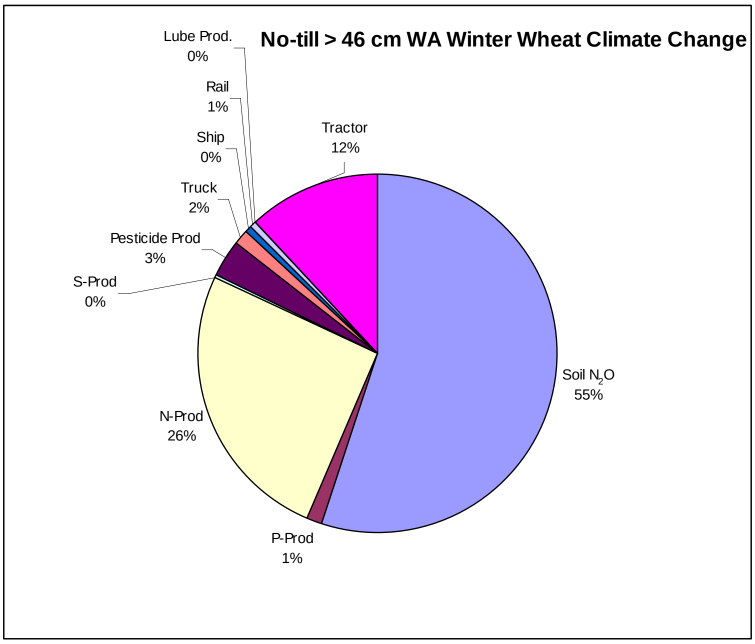

Fig. 4: Climate change contribution analysis no-till winter wheat

Tab. 1: Modeled carbon sequestration in Washington wheat rotations

Wheat Crop Rotation	Sequestered Carbon	N_2O Losses	Net GHG Emission
	Kg CO_2/ ha-year		
Lind CT	100	187	87
Lind RT	200	327	127
Saint John CT	200	233	33
Saint John RT	275	117	-158
Pullman CT	430	1493	1063
Pullman RT	440	1634	1194
Pullman NT	480	653	173
Pullman CT(Peas)	90	1260	1170
Pullman RT (Peas)	150	933	783
Pullman NT (Peas)	130	373	243
Othello Representative	750	1634	884
Othello Reduced Tillage	650	1120	470
Othello Minimum Tillage	750	1984	1234
Mean	**357**	**919**	**562**
S.Deviation	**243**	**646**	**508**

The Climate Friendly Farming Program has modeled the sequestration of carbon into soils using the Crop-Syst[i] program, and compared them to the calculated N_2O emissions over the range of tillage and rainfall scenarios. Tab. 1 shows that in general, the amount of both carbon sequestration and of nitrous oxide emissions are very variable, but that in only one reduced tillage case does it appear that the overall system provides a net greenhouse gas sink. This one case may be a statistical error, since it represents one of 16 modeled systems.

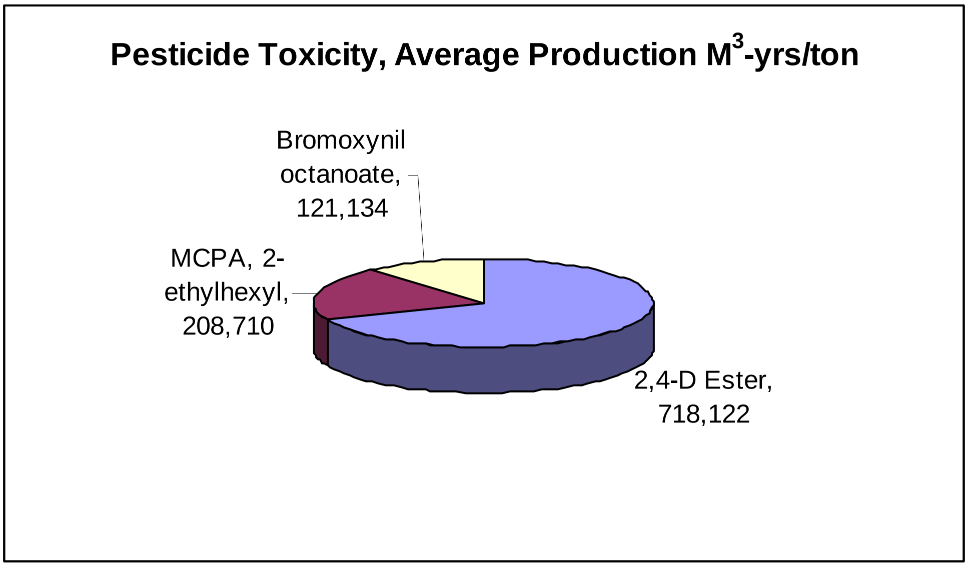

Fig. 5: Pesticide toxicity average winter wheat

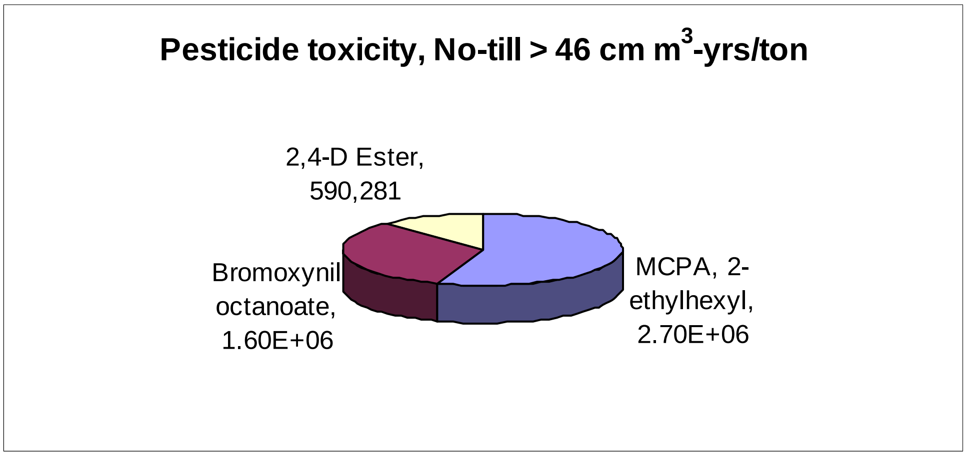

Fig. 6: Pesticide toxicity, no-till winter wheat

In analyzing the option of growing wheat with no-till, we calculated the ecotoxicity of growing average wheat versus no-till wheat grown in relatively high rainfall areas. The method used was that of Schenck (2007), which is an automated variant of the SETAC-UNEP USE-tox method, in which ecotoxicity, bioaccumulation and persistence are automatically calculated rather than measured using the USEPA PBT profiler, US EPA(2006). This approach allows the characterization of any substance for which either the Chemical Abstracts number or the physical structure are known.

Fig. 5 and 6 show the ecotoxicity of the pesticide application for approximately 99% of the calculated toxicity. The three same herbicides are shown as dominating the pesticide toxicity: 2,4-D ester, Bromoxynil and MCPA. The latter two pesticides are applied simultaneously. The total toxicity of the conventional use is 1.1 million m^3-years per ton of wheat, substantially less than the 4.9 million m^3-yr per ton grown under no till conditions (>46 cm rainfall).

Discussion

Four questions were addressed with this study: 1) can carbon be sequestered in soils even while crops are being produced, and 2) is that sequestration large enough to offset the climate change effects of life cycle greenhouse gas emissions? 3) No-till systems reduce fossil fuel consumption but what is the effect on the higher herbicide use they require? Finally 4) what advice can be given to farmers to reduce the overall environmental impacts of growing wheat?

The majority of the climate impacts of wheat production in Washington State appear to be incurred at the farm, rather than upstream. High emissions of nitrous oxide dominate the carbon footprint in all cases, but this is especially stark in the case of no-till systems, where fossil fuel use on-farm is minimized. Although all tillage systems act to increase carbon in the soils, the emissions of N_2O are much larger in terms of CO_2 equivalents. In these systems, carbon sequestration in soils is palliative with respect to greenhouse gases, but overall, the agricultural systems emit much more global warming potential than they sequester, even in low-till systems (with one possible exception).

What is very clear is that the nitrogen biogeochemistry is driving the climate change effects of farming wheat. Further research into methods to limit nitrous oxide formation is underway. Until best practices can be developed the best advice one can give farmers is to reduce tillage: this yields economic and soil conservation advantages as well as climate change improvements.

The use of herbicides in both conventional and reduced tillage scenarios provided a very large amount of ecotoxicity, driven by only a few herbicides. Choosing less-toxic herbicides can have a near-term effect, and farmers should be encouraged to do so right away. The Cooperative extension service in Washington State is appropriately placed to take on this issue.

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Strawberry and tomato production for the UK compared between the UK and Spain

Adrian Williams[1], Emma Pell [2], J Webb [2], Ed Moorhouse [3] and Eric Audsley [1]
[1] Natural Resources Management Centre, Cranfield University, Bedford, MK43 0AL, UK. [2] AEA Energy & Environment, Gemini Building, Didcot, OX11 0QR, UK [3] Moorhouse Consulting, Summerhill Cottage, Cranbrook, TN17 2AR, UK.

adrian.williams@cranfield.ac.uk

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Abstract

Comparative Life Cycle Assessments (LCA) were made between the production of strawberries and tomatoes in Spain and the UK. The functional unit was 1 t crop, packaged and delivered to a retail distribution centre (RDC) in the UK. One grade of strawberries was considered, but tomatoes were sub-divided into loose classic, classic on-the-vine and loose baby-plum.

Production of both tomatoes and strawberries in Spain took less energy than in the UK and incurred lower potentials for global warming and acidification. The main term in the UK for tomato production is energy for heating and electricity, but energy savings of about 90% were possible by using waste heat and CO_2 and about 30% by using combined heat and power (CHP). The burdens of high yielding loose classic tomatoes are lower than those sold on the vine or loose baby plum tomatoes, because they are lower yielding. Transport from Spain alone did not compensate for the higher energy inputs of UK tomato production.

Strawberry production burdens were relatively more mixed. The lower energy of Spanish production was coupled with higher transport energy so that total energy was about the same as UK production. Other relative burdens of strawberry production were more varied, e.g. eutrophication potential much higher in Spain.

Water for irrigation is a cause for concern in southern Spain.

Introduction

The UK climate does not support the growth of crops, like tomatoes, outside their natural outdoor growing season without growing them in protected environments and sometimes supplying heat and /or light. The cropping season for fruits like strawberries has been extended by breeding varieties that bear fruit through the summer (ever-bearers) and using unheated polyethylene covered tunnels (polytunnels). These techniques have extended UK strawberry cropping to be from May to October. The use of heat (and the associated CO_2) has extended tomato cropping to a season of about March to October, but with large energy needs. There is a desire to eat fresh fruit and salad crops for most of the year and alternative sources have been developed in countries around the Mediterranean especially Spain. Spanish producers have concentrated on supplying fresh produce in the winter months, but the seasons also overlap in the autumn and spring. This study compares tomato and strawberry production in the UK and Spain. Because of the seasonal complementarity, the products are only available for a limited period, so this study has developed into one in which the maintenance of a seasonal supply is maintained, with limited duplication.

Method / Approach

Functional units

The functional unit for both crops was 1 t delivered to the regional distribution centre (RDC). Distinction was made between different types of tomatoes as these have different burdens, i.e. loose classic, vine-marketed classic and lose baby plum.

Tomato production systems

Tomato production was modelled using the same basic approach as taken by Williams *et al*. (2006), which included producer data from growers, which were considered to be representative of the industry as a whole. Data were also obtained from a production centre in Almeria in south eastern Spain. New inventories were created to deal with the water supply, which included some desalination and reservoir water in Spain as well as some fertilisers that were specific to Spain. Burdens of electricity and other fuels were derived from the EU's European Platform on LCA (EU, 2008) for both Spain and the UK.

In Spain, tomatoes are overwhelmingly grown along the SE coastal strip from Malaga to Alicante, including the main greenhouse area to the East and West of Almeria. The cheaper tomato products (loose classic) are often grown in basic polyethylene tunnels, while the more sensitive products (e.g. baby plum and on-vine) tend to be produced using more sophisticated growing systems and are more comparable to typical UK production systems, albeit in polyethylene clad houses rather than glasshouses.

In the UK, greenhouses are mostly heated with natural gas, either by stand-alone boilers or combined heat and power (CHP) units. A few sites have been developed where local, industrial waste heat and CO_2 is used, e.g. next to a Sugar factory in Norfolk and next to a nitrogen fertilizer plant on Teesside.

Heat is required to provide an optimal growing environment throughout the year with obvious peak demands during the winter and early spring. Some heating is used to reduce humidity levels and thus to avoid condensation, so helping to minimise fungal diseases (so reducing the need for fungicides). The CO_2 from combustion is fed into glasshouses throughout the growing period to enhance photosynthesis and crop productivity. The addition of CO_2 is most critical in the summer when the crop is most actively photosynthesising and ambient CO_2 concentration can be depleted and so limit photosynthesis and hence yield.

A minority of glasshouses use lighting to enhance growth in the darker months, but this currently represents a small part of the UK business and was not included in this analysis.

Different approaches are needed to analyse the alternative sources of electricity, heat and CO_2. The stand-alone boiler method simply requires the amounts of gas and electricity to be known. For CHP, the most common way is to consider the gas used in the house and calculate a credit for the exported electricity, which is based on a comparison with the most modern generating method. The CHP system that we consider is a unit that burns gas, e.g. a reciprocating gas engine of 1 MW generating capacity per ha, where the heat and CO_2 are used for tomato production as though from a stand-alone boiler. The internal heat and electricity needs of the glasshouse are met by the CHP unit and most the generated electricity is surplus and hence exported to the national grid. Because so much electricity is exported from such systems, the analysis is based on comparing conventional electricity generation for the national grid with generation by CHP in which tomatoes are a co-product. A combined cycle gas turbine, CCGT, (Defra, 2008) is used for the analysis, and it represents a comparison of new marginal generating capacity rather than the existing generating mix.

When using "waste" heat and CO_2, the analysis depends on how truly that heat would otherwise be wasted. If the industrial process is not compromised in order to supply the heat and CO_2, then the comparison needs to account only for the extra burdens of delivering the heat and CO_2 to the greenhouse (plus any extras needed to distribute it more than in a standard house). It is our understanding that the heat and CO_2 is a genuine waste. The additional burdens of supplying heat and CO_2 are based on the pipe lengths and diameter, and fluid flow rates, hence the pressure drops and power needed for pumping. An allocation for pipe materials and installation was also made

The sources of heat and CO_2 have a major impact of the burdens of growing tomatoes in the UK. The proportions in Tab. 1 are based on an estimate of 35% of tomato growers of 150 ha using CHP and the two known producers using waste heat and CO_2. The results of the comparison with Spanish production are based on this assumed industry mix. The benefits of CHP and waste heat and CO_2 use are shown later.

Tab. 1: Energy and CO_2 sources in long-season UK tomato production

Type of heat source	Proportion by area
CHP	35%
Waste heat and CO_2	13%
Stand-alone boilers (mainly natural gas)	52%

Post farm gate

Data on transport and packaging were derived from the UK national emission inventories and the *Ecoinvent* database. Data were identified for post farm gate handling activities, included location of packaging manufacture and hence delivery distance to the producers, packaging and initial cooling energy inputs and the distances between the production site and RDC. A central location was assumed for the UK of Corby, which is the economic geo-centre of the UK.

Strawberry production systems

The UK strawberry season lasts from about April to October, with the peak supply in June, owing to crop historically peaking in June, hence June bearers. Longer season cropping has been made possible with ever-bearer varieties and the use of polytunnels, together with phased planting of June bearers to enable their first cropping season to occur outside of June, although reverting subsequently. There are about 14 main production systems in the UK (UoH, 2007), increasing to 21, including sub-systems. Variations include: growth medium, crop variety, planting time, years of cropping (one to three), polytunnel use (for some or all of the crop life) and the use of soil fumigation (Tab. 2). There is some organic production, but this was not included in the analysis.

The main bio-physical characteristics of the production systems were taken from the UoH report (UoH, 2006) together with data from the Pesticide Usage Survey (Garthwaite & Thomas, 2003) and all were interpreted on the basis on long term crop-soil balances (Williams *et al*., 2006), e.g. offtake of P and K in crops plus losses must equal the long term supply.

Tab. 2: Main features of strawberry production in the UK

Protected	75%	Unprotected	25%
Soil	85%	Substrate	15%
Raised bag	50%	Table bag	50%
Coir	25%	Peat	75%
Fumigated	85%	Non-Fumigated	15%
Spring planted	30%	Summer planted	70%
June bearer	50%	Ever-bearer	50%

Industry specialists were consulted to estimate the proportions of production systems in use (Tab. 2). This was challenging because strawberry production is in a state of rapid flux. One reason is the ban on the use of methyl bromide (MB) as a soil sterilant. This product played an important role in crop production systems, but it was also a major pollutant and its use was phased out under EU regulations from 2005 (Regulation (EC) No 2037/2000). This has accelerated a move away from soil-based production. The development of production systems has also been driven by other factors in the UK, primarily labour costs and crop quality. Table-top systems are increasingly being used by growers and because they deliver improved crop quality, reduce labour costs and offer better physical welfare for workers, but they incur higher burdens for the materials used in construction. The addition of protective tunnels has also resulted in more reliable production with substantial quality improvements and reduced pesticide usage. The environmental burdens associated with the national crop are greatly influenced by the changes, both between and within, the various production systems and it is difficult to present Life Cycle Inventory (LCI) values that will be reliably long lasting because of the rate of innovation and the range of techniques used by growers.

Most Spanish strawberries that are supplied to the UK market are grown in the Huelva area in SW Spain. Strawberry production dominates this area and this presents some significant production and environmental challenges. Most crops are grown in what are effectively annual mono-cultures, with extensive use of soil fumigation, which was based on MB, but is now being replaced by other chemical treatments, or moving to container production. Polyethylene clad tunnels are used for protection on most crops. About 90% are micro-tunnels (rather like elongated cloches) and 10% macro-tunnels (that are similar to those in Britain, which are known as Spanish tunnels). Most are grown in sandy soils, which leach readily. Polyethylene film is used extensively for mulching, soil fumigation and solar disinfection. One difficulty is that the soil fumigation options that are being adopted by Spanish producers to replace MB are still being developed and the applied quantities of chloropicrin, metam-sodium and metam-potassium, the main alternatives, are somewhat uncertain. We used typical UK application rates per ha. Spanish producers had an exemption from the ban on MB until December 2007 (2005 in the UK) and it is not clear if all stocks in Spain have been used. This means that results, even from recent studies, may not be wholly representative of exactly what is happening now due to the on-going changes within the industry. The phasing out of MB seems likely to move production increasingly towards substrate production in containers or bags with peat and/or coir as the main substrates. The move to substrate production is more advanced in the UK than in Spain and the bulk of the Spanish crop is still soil-grown. Only soil-grown systems are reported here. We also results assuming no MB is used, but show how its use would have affected production burdens.

Data sources

The main sources for Spanish production have been the scientific literature, data from Spanish web sites, including the national and local governments (e.g. the specification for integrated production) together with limited data from producers and the Spanish branch of the WWF. Unlike the UK, Spanish production techniques appear to be fairly uniform with about 90% production in micro-tunnels and almost all production currently in soil.

Caution

It was difficult to obtain data from producers both in the UK and Spain and so the results should be viewed with some caution. Various simplifying assumptions were necessary, e.g. Spanish pesticide use per ha was assumed to be as the UK average, which is not what anecdotal evidence might suggest. There are thus some areas of considerable uncertainty and we also clearly recognise that the industry is in flux. We consider it reasonable to consider the findings as indicative, but not necessarily definitive. Future trends could also quickly change the industry structures (e.g. more container growth) and we do not know what effect the move from MB will have on long term Spanish soil-based production.

Another area where data sourcing proved impossible was young plant production. It is reasonable to expect the methods to be similar in both countries. We know that plants are often refrigerated before planting to manipulate crop timing. Given that most Spanish producers grow the crops for one year only, the plant overhead per ha is probably larger than that in the UK where two years of cropping is closer to the national average. In contrast, the Spanish production yields more per ha, so the actual difference per t is probably small and its omission should not have a major impact on the final result.

Seasonality

Spanish production is targeted mainly towards the UK off-season from February to May. The UK season now covers March to October, but with a clear peak in June. There is clearly some overlap in spring, but much of the production fills complementary seasons. There is also a high consumer demand for the seasonal UK strawberry crop and a perceived quality advantage of the UK crop.

Results

Tomatoes

The results for tomatoes show that burdens increase as tomatoes get smaller and if grown for on-the-vine sale rather than loose. This is readily explained in the UK where energy inputs in standalone boiler-heated houses account for about 97% of energy use and GWP. The specialist tomatoes are lower yielding so the burdens increase. Specialist tomatoes do, of course, offer consumers a better sensory experience than classic loose tomatoes, partly because they are more mature when picked. This means that classic loose need less packaging. In Spain, the yield differences still have some effect, but the biggest change is between baby plum and vine classic, because heat is used in the more sophisticated houses used for the highest value crops. In this case, propane was the fuel, which appears to be associated with much higher emissions of compounds causing potentials for ozone depletion and photo-chemical oxidation, which is why these numbers are so much higher. There was a universal trend for burdens to increase; from loose classic through vine classic to baby plum, reflecting that higher economic cost is also associated with high environmental cost for tomatoes.

Burdens of energy use, GWP, acidification potential and abiotic resource use were larger in the UK than Spain. This resulted from the much greater use of energy in the UK in primary production and this always outweighed the extra transport and cooling needed to deliver Spanish tomatoes by road to the UK. The ratio of UK / Spanish burdens decreased from loose classic through vine classic to baby plum reflecting the higher inputs in Spain for the more specialist types.

The other burdens were greater for tomatoes from Spain for a mixture of reasons. For example, the use of propane for heating the house for baby plum tomatoes gave higher ozone depletion and photo-chemical oxidation potentials in Spain, which would probably not have occurred if natural gas was used. The higher eutrophication potentials in Spain are mainly a consequence of growing the crop in sandy soil, while the UK crops are all grown in recirculating nutrient solutions.

Primary production dominates most burdens both in the UK and Spain, but with more exceptions for Spanish production (Tab. 4). The exceptions are mainly related to the long transport stage (about 2400 km), so that the lowest pre-farm gate proportions of burdens such acidification from Spanish tomatoes occur with loose classic tomatoes, which have the lowest pre-farm gate burdens.

Tab. 3: Burdens of tomato production in UK and Spain and delivery to the UK RDC (per t crop)

	UK			Spain		
Burden	**Classic loose**	**Classic vine**	**Baby plum**	**Classic loose**	**Classic vine**	**Baby plum**
Primary energy, GJ	36	83	95	8.7	14	45
GWP, kg CO_2 eqv.	2.2	5.1	5.9	0.74	1.0	3.1
Eutrophication potential, kg PO_4 eqv.	0.21	0.29	0.28	0.47	0.49	0.71
Acidification potential, kg SO_2 eqv.	2.4	4.4	4.9	4.6	5.5	10
Ozone depletion potential, kg CFC-11 eqv.	0.50	1.0	1.0	0.78	1.85	29
Pesticides used, kg A.I.	0.29	0.70	0.81	2.2	2.2	1.5
Abiotic resource use, kg Sb eqv.	18	41	48	14	25	39
Land occupation, m^2	19	44	51	89	130	190
Irrigation Water, m^3	24	58	67	36	51	70
PM_{10}, kg	0.11	0.13	0.14	1.1	1.5	2.1
Photo-chemical oxidation potential, kg ethylene eqv.	0.10	0.21	0.25	0.13	0.17	2.05
Non-methane volatile organic carbon, kg C eqv.	0.25	0.40	0.44	0.67	0.77	3.1

Tab. 4: Proportions of pre-farm gate burdens

Burdens	UK			Spain		
	Classic loose	Classic vine	Baby plum	Classic loose	Classic vine	Baby plum
Primary energy used, GJ	94%	98%	98%	51%	70%	91%
GWP, t CO_2 eqv.	94%	98%	98%	36%	59%	86%
Eutrophication potential, kg PO_4 eqv.	47%	68%	68%	31%	39%	57%
Acidification potential, kg SO_2 eqv.	57%	75%	78%	36%	55%	75%
Abiotic resource use, kg Sb eqv.	92%	96%	97%	76%	86%	91%
PM_{10}, kg	26%	52%	57%	84%	90%	93%
Photo-chemical oxidation potential, kg ethylene eqv.	81%	94%	93%	54%	61%	97%

Alternative sources of heat and CO_2

Standalone boiler systems use much energy and the benefits of using CHP and waste heat and CO_2 are large (Fig. 1). Using CHP reduced energy consumption and GWP by about 30%, while using waste heat reduced energy consumption and GWP by about 90%. The energy needs for baby plum tomatoes produced in the UK with waste heat and CO_2 were even below the Spanish energy needs.

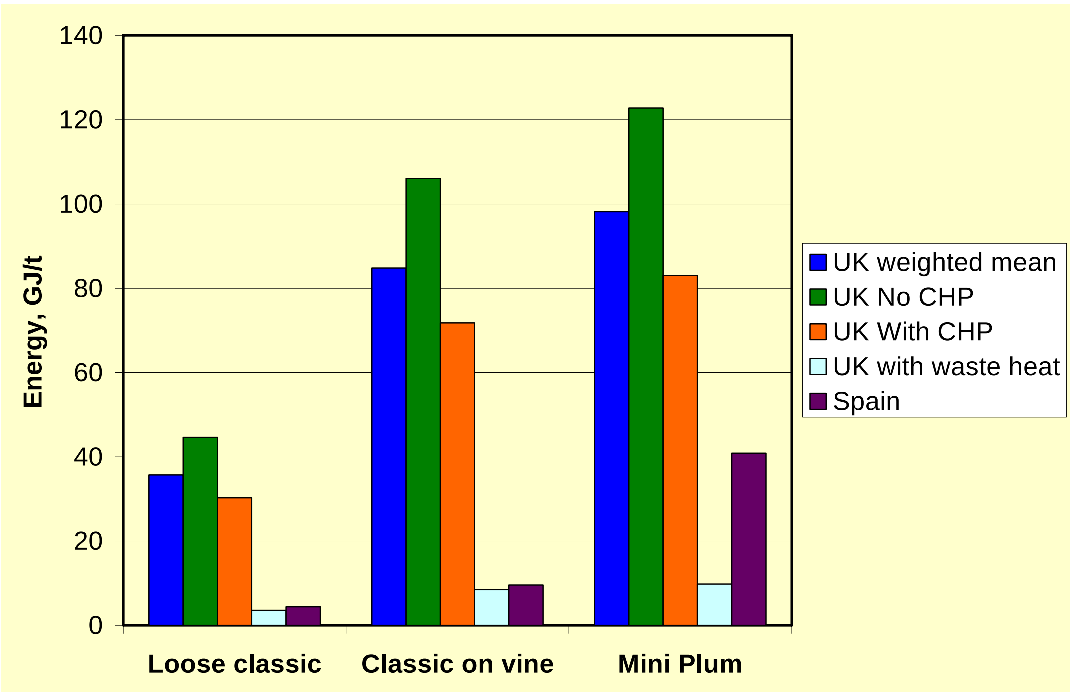

Fig. 1: Energy use for tomato production in Spain and the UK both with and without CHP or waste heat and CO_2.

Strawberries

The burdens (Tab. 5) for strawberries at the RDC are an interesting mixture. The higher energy use, acidification potential and GWP for UK strawberries are roughly equal to the higher delivery burdens from Spain so that these totals are about the same. Eutrophication is noticeably larger in Spain owing to high N fertiliser applications, sandy soil and higher irrigation rates per ha. Land occupation in Spain is lower as the yields are higher than those in the UK. Pesticide use also appears much lower in Spain, but it must be remembered that actual values for Spain were not obtainable, so the same application rates per unit area as in the UK were assumed, so they are actually a function of yield.

About 80% of the post farm gate burdens in the UK are from packaging manufacture, mainly the punnets themselves. This is similar in absolute terms in Spain, but proportionally lower owing to large transport burdens from Spain.

Tab. 5: Main burdens or producing strawberries in the UK and Spain. All results are shown per t to the farm gate without packing. These results assume that no MB is used.

	UK			Spain			[Total Spain] / [Total UK]
Burden	**pre-FG**	**post-FG**	**Total**	**pre-FG**	**post-FG**	**Total**	
Primary energy used, GJ	13	1.5	14	8.3	4.4	12	87%
GWP_{100}, t CO_2 eqv.	850	140	990	350	560	910	91%
Eutrophication potential, kg PO_4 eqv.	2.5	0.094	2.6	15	0.40	15	600%
Acidification potential, kg SO_2 eqv.	6.5	1.3	7.7	3.9	3.2	7.1	92%
Ozone potential depletion, g CFC-11 eqv.	3.0	ND		1.5	ND		
Abiotic resource use, kg Sb eqv.	13	2	15	3.7	3	6.7	45%
Land, m^2	54	NA	54	26	NA	26	48%
Irrigation Water, m^3	110	NA	110	130	NA	130	
PM_{10}, kg	ND	0.079	0	ND	0.22		
Photo-chemical oxidation potential, kg ethylene eqv.	0.59	0.02	0.61	0.16	0.075	0.24	39%
Non-methane Volatile Organic Carbon, kg C Equiv	1.8	0.16	1.9	0.66	0.50	1.2	63%
Proportion of renewable primary energy, %	5.8	1	6	7.1	2	5	

NA = Not applicable, ND = Not determined.

If MB was still being used in Spain, the total energy and GWP would be about 10% higher, but ozone depletion potential would be about 1,700 times higher. This highlights the harm MB does and illustrates well the reason for its ban by the EU.

Discussion

The results were contrasting between crops. The energy used for growing all crops was lower in Spain, relatively more so for tomatoes than strawberries owing to the heat and electricity used in UK glasshouses. Post farm gate burdens were similar between crops (about the same distance and broadly similar amounts of packaging). These were proportionately higher for loose classic tomatoes and strawberries than vine classic or baby plum tomatoes.

Our estimates of GWP for UK and Spanish production of loose tomatoes (2.2 and 0.74 t CO_2-equiv per t tomatoes respectively) are very similar to those of Smith *et al.* (2005) of 2.4 and 0.63 t CO_2-equiv per t tomatoes. Biel *et al*. (2006) compared tomato production in Denmark, the Netherlands and Sweden up to the RDC in Sweden. Their GWP values were also similar to our estimates for the UK at 3.6, 2.9 and 2.7 2.72 t CO_2e per t in Denmark, the Netherlands and Sweden respectively, with production being the clearly dominant phase. Their values for energy use were I the range 1.4 to 1.7 times our UK ones, but it is not clear how much, if any, CHP was used in their analysis.

The benefits of using CHP and, in particular, waste heat and CO_2 in UK tomato production are considerable, with latter making UK production even more energy efficient than some tomato production in Spain (when heat is used in Spain). There is nothing in principle to stop Spanish producers making use of such energy sources. Re-location of current standalone boiler systems in the UK production to be near sources of waste heat and CO_2 seems to be a rational policy option, but a major constraint is the availability of flat land near to such sources. An additional caveat is that the sources should be genuine waste.

Our estimate of 0.7 t CO_2-equiv per t strawberries produced on UK farms falls between the estimates of Lillywhite *et al*. (2007) 1.2 and UoH (2005) at about 0.4 t CO_2-equiv per t strawberries. Their analysis made different assumptions about the use of polyethylene tunnels etc, and used different inventories for data, but the general agreement is very encouraging.

Both Spanish strawberry and tomato production used more water than in the UK, which was also associated with more eutrophication in Spain. This has long term consequences because Spanish water takes more energy to deliver because techniques like reverse osmosis desalination are increasingly required as aquifers have become polluted and “fossil” water reserves depleted. This phenomenon cannot be attributed wholly to these crops, as other crops are grown, together with increased direct human consumption.

The phasing out of methyl bromide as a soil disinfectant has clearly reduced the ozone depletion potential of strawberry production, particularly in Spain where its use was more widespread owing to the annual mono-culture approach to growing strawberries. The future trends in production are likely to be towards more container production as well as growers becoming familiar with the effectiveness of other soil disinfection techniques. Time will tell how yields and production burdens will change, but they could be very different in a few years time. One curiosity is that strawberries were so named because they were grown on straw-covered soil, but straw is hardly used now.

The overlap in seasons is relatively short for both crops and during those periods the comparisons are valid, but the results show what is more like an extension of the supply season in the UK. The main alternatives for tomatoes in high summer are other northern European countries with similar heated production systems, because it becomes too hot for Spanish producers. Some caution is also needed because the crop qualities and varieties are not identical. UK producers and suppliers maintain that the UK produce is of better quality than that from Spain.

The comparisons show mixed effects of growing crops out of the UK’s natural growing season. A problem that is common to all crops is the increasing environmental cost of supplying water in areas that have naturally low rainfall. While global warming is often assumed to be of prime concern, this resource limitation cannot be overlooked and it seems likely to increase in importance this century, especially in the Mediterranean basin.

Conclusion

Production of both tomatoes and strawberries in Spain took less energy than in the UK and incurred lower potentials for global warming and acidification, but the seasons have a limited overlap and are more complementary than competitive.

The main term in the UK for tomato production is energy for heating and electricity, but energy savings of about 90% were possible by using waste heat and CO_2 and about 30% by using CHP.

The burdens of high yielding loose classic tomatoes are lower than those sold on the vine or loose baby plum tomatoes, because they are lower yielding.

The differences between baby plum production in Spain and the UK were much lower than for other tomato types because they are of higher value and thus heat is used for their production in Spain.

Land use, eutrophication potential and irrigation water use were generally higher in Spain than the UK for tomato production.

Transport from Spain alone did not compensate for the higher energy inputs of UK tomato production.

Strawberry production burdens were relatively more mixed. Lower energy for Spanish production was associated with higher transport energy so that total energy was about the same as UK production.

Other relative burdens of strawberry production were more varied, with eutrophication potential much higher in Spain, but abiotic resource use lower.

There are qualitative differences between the produce from Spain and the UK, with a UK perception of higher quality of UK produce.

Water use for irrigation in southern Spain is a major problem. The environmental cost of water has increased through pollution of groundwater and the need to provide more infra-structure to supply water. This appears to be a growing problem in the Mediterranean basin.

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LCA as environmental improvement tool for products from line caught cod

M. Vold[1] and E. Svanes[1]
[1] Ostfold Research, Gamle Beddingvei 2b, N-1671 Kraakeroy, Norway, mie@ostfoldforskning.no

Keywords: LCA, autoline fisheries, cod, resource efficiency, product quality, value chain assessment

Abstract

A Norwegian research project “from Seafloor to Consumer” (www.bunntilmunn.no) aims at reducing the total environmental impact of fish consumption by demonstrating the quality and environmental performance of line-fished cod and identifying, documenting and implementing improvement measures.

LCA has been conducted on products derived from the main outputs of cod processing: Loins, portions and mince/block. The fish is caught by autoline fishing vessels that operate in the North Sea and the Barents Sea. The LCA shows that the impact from the fishing phase is the dominating environmental impact. However, the impacts from the other life stages are considerable, as demonstrated by the studies by Svanes et al, 2008; Thrane M, 2006; Ziegler et al, 2003 and Liodden et al, 2003. This LCA study clearly shows that the focus of improvement measures in the other life stages should be to increase the yield and reduce product loss.

The LCA is used together with other scientific tools such as quality analyses, monitoring of conditions in the value chain as well as assessment of priorities in the market, to identify and choose the best options for improvement.

The LCA identified leakage of cooling agent in the fishing boat freezing system as one dominant GHG emission source that could readily be remediated. This was new knowledge for the fishing boat operator and led to a decision to replace with a natural cooling agent.

In an effort to achieve environmental improvements and demonstrate the environmental performance of the products the production company and fishing company have applied for, and received ecolabelling certification for some of the products sourced from autolinecaught cod. The certification is according to both the Marine Stewardship Council (MSC) and the Swedish Ecolabel KRAV.

Introduction

In the project;”From Seafloor to Consumer” different companies and research institutes have joined forces to cover the whole value chain for line caught fish from the catch to the consumers purchase.

Several studies (Ziegler, 2001; Thrane, 2006 and Eyjólfsdóttir HR, 2003) indicate that passive fishing methods such as autoline and Danish Seine is more energy efficient than active methods such as trawling.

The main goal for the project is to contribute to increased environmental- and resource efficiency in the fishing sector and to give consumers a better access to ”clean Norwegian food” of high quality. Basis for the study the value chain of line caught fish and development of new and more efficient long line equipment for small fish boats. This paper describes how environment- and resource assessment of a reference system are used to identify the important areas for improvement.

Method / Approach

LCA is used together with other scientific tools to identify and choose the best options for improvement from a holistic view. The value chain includes all steps from the production of fishing gear to consumer. This is a complicated value chain with different species, processes and products included. The value chain is schematically described in Fig. 1.

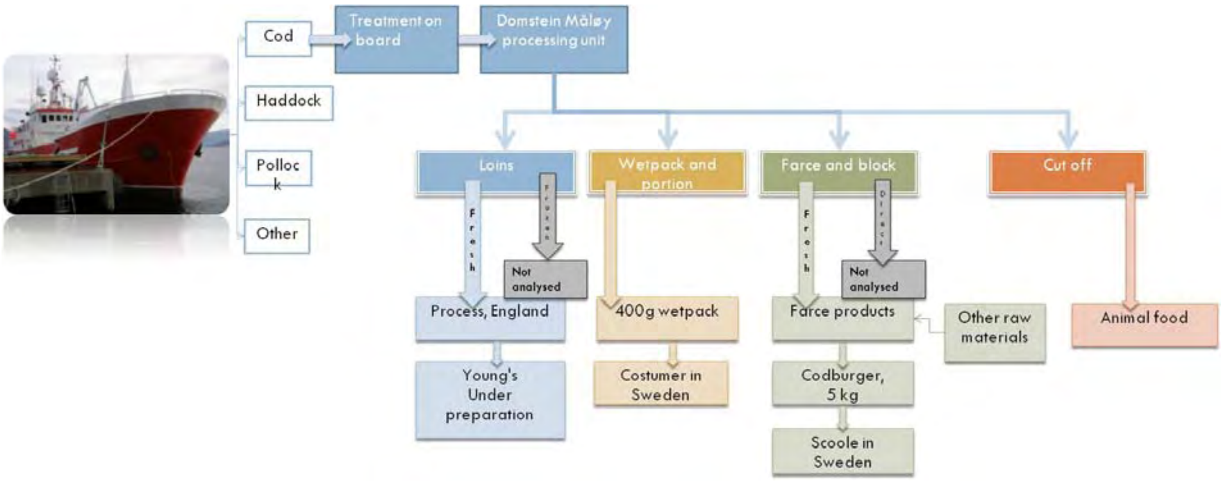

Fig. 1: Product system for line caught fish.

The study has not investigated the last life cycle stage, namely what happens after the product is purchased. Several studies have shown that a large proportion of household waste in the Nordic Countries is food. Part of it is not fit for consumption such as orange peelings but other parts are consumable but thrown for other reasons (too much food prepared, too much bought, etc). How much of these cod products are used by the consumers? We do not know but a small survey of cooks and serving personnel in Swedish schools indicate that the wastage is very small. This is mainly motivated by the high price of the products. One might expect the consumers to act in the same way.

LCA

An LCA of a product is defined as a systematic mapping and assessment of environmental and resource impacts throughout the entire life cycle of the product. The LCA methodology includes all processes and activities that are part of a product system, and thus contribute to achieving the function or functions that the product system shall fulfil (ISO 14044).

This project had emphasis on getting site-specific data. Data collection was performed through on-site inspections, questionnaires and interviews, in close collaboration with the participating parts in the project.

1. Between kinds of fish in each catch (In Norway all caught fish has to be taken ashore).
2. Between different parts of the fish which are used for different products.

Economic allocation has been used for all. According to one study (Ayer et al, 2006) this is the most frequently used allocation method in LCA studies of fish, even though ISO recommends system expansion to avoid allocation as first option and physical causality as second option. Economic allocation was chosen because it reflects the priorities of the actors in the value chain and encourages a high yield of fish for human consumption. System expansion to avoid allocation between different fish species in the catch, as done by Thrane, 2006, was not feasible because data is lacking for fisheries targeting each of the bycatch species. System expansion to avoid allocation in the processing stage (also according to Thrane, 2006) was not done because it is difficult to identify any “avoided product”. It can be discussed if other allocation methods would have been more relevant if a profile of the life cycle was the goal for the project (E.g. if the goal was to compare a cod product with a chicken product) The choice of allocation method seems to be of minor importance since the goal in this stage is to find options for improvement in one value chain.

Market Priority Assessment

Domstein ASA has over the years done a significant effort to reduce the environmental impact of their products. These efforts are still going on. The company has noticed an increasing scepticism in the general public towards cod products because of negative publicity, especially in Swedish media. Many environmentally conscious people have advocated a stop in the cod consumption. This is largely due to the difficult situation for the Baltic Sea cod, whose stocks are severely depleted. Domstein has not been able to adequately communicate the good environmental performance (and high quality) of products derived from autolinecaught cod from the Northeast Arctic stock. Hence the company took the strategic decision to let independent organisations document the products high environmental performance through well-established tools, namely the MSC and KRAV ecolabels.

These labels focus on the biological sustainability of the fish stocks but also take into account other environmental impacts through a number of requirements. KRAV is more comprehensive but little known outside Sweden. MSC is internationally acknowledged. Both labels as regarded among consumers as neutral and trustworthy.

Improvement analysis

LCA was used together with other scientific tools to identify and choose the best options for improvement. All methods give important areas for improvement. Earlier experiences show that improvement in one area can cause problems in others. By holding the results together, compare and analyse, one will obtain a more holistic view than by the different separately.

Results

The LCA clearly identified the fishing and processing stages as the environmentally most important steps of the Life Cycle. Fig. 2 and 3 show some examples of value chain assessments for one product in the project (400 g Wetpack Cod). The reason that we have chosen to focus on the results for wetpack is that this product is the most representative product. It is a frozen product and lies in value in between the loins products and the products from mince and block. Furthermore the wetpack is only processed once and only consist of cod. The cod burgers and the other deep-fried products consist only partly of cod and are processed twice (first cutting, then mixing and frying).

The catch also was seen as most important in the market priority assessment.

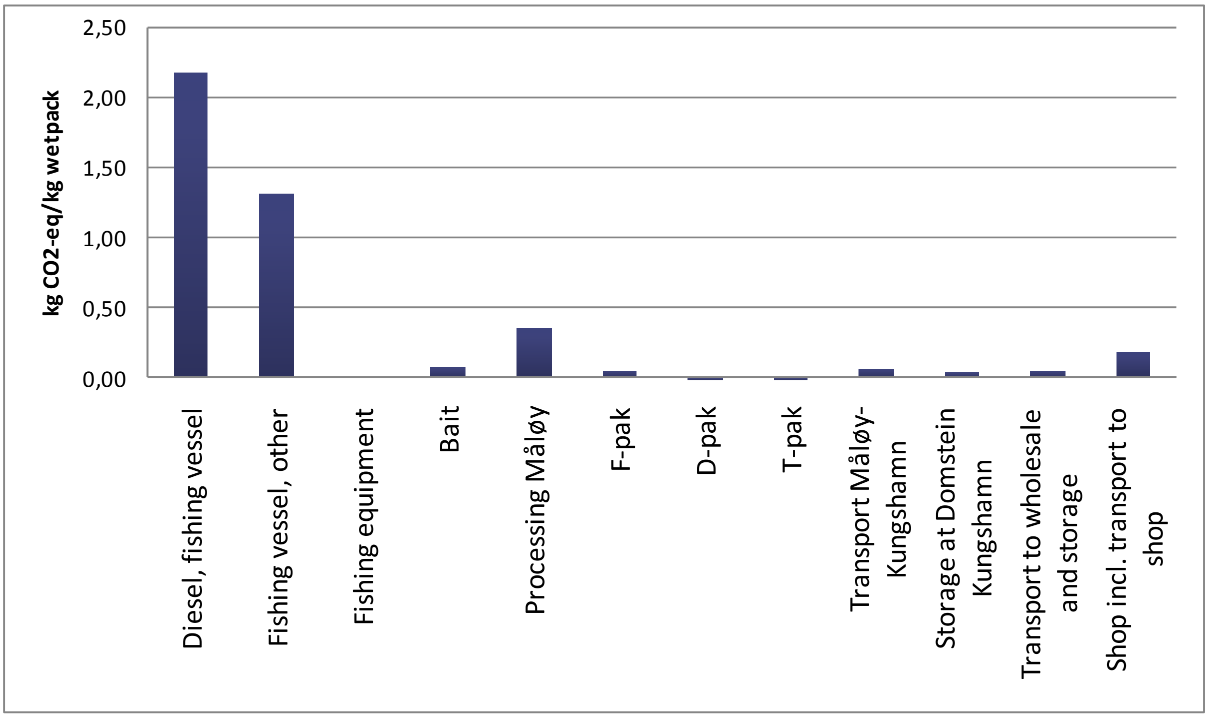

Fig. 2: GHG emissions from seafloor to shop for the product 400 g Wetpack, cod.

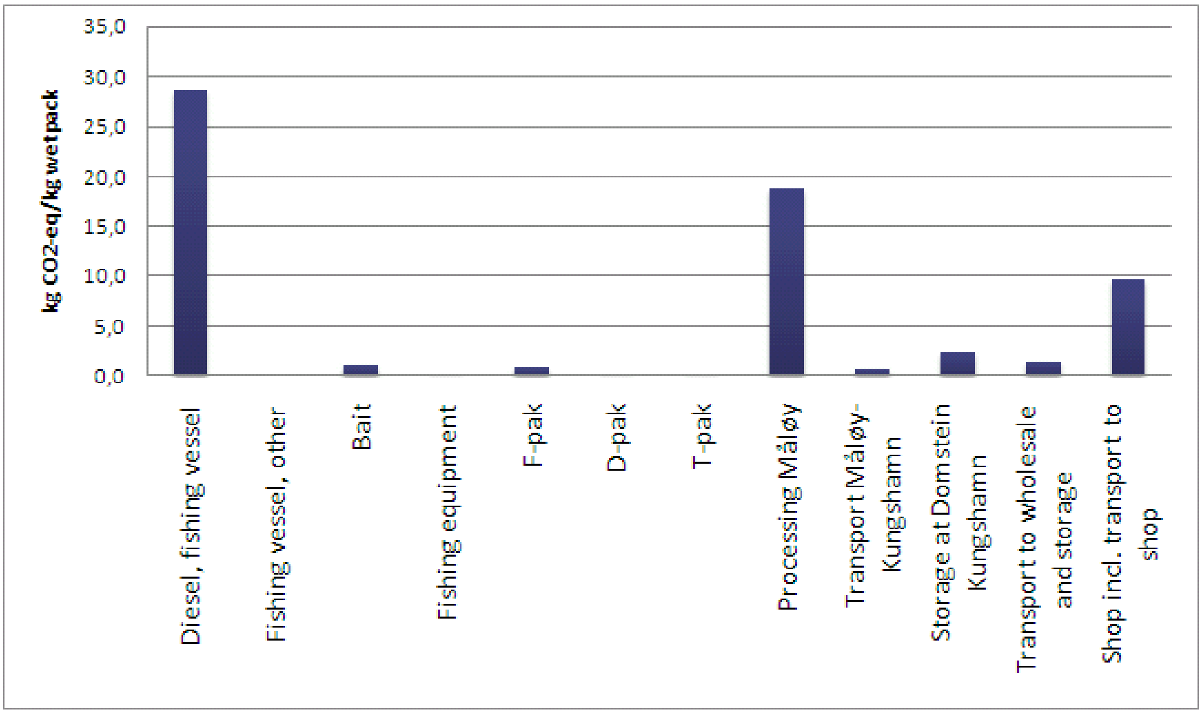

Fig. 3: Energy consumption from seafloor to shop for the product 400 g Wetpack cod.

Fig. 2 and 3 describe GHG emissions and primary energy consumption through the value chain. The GHG emissions stem largely from the fishing boat, with two major sources, consumption of fossil fuel and leakage of cooling agent. The energy consumption, on the other hand, is substantial also in the parts of the value chain. The other impacts studied are photochemical oxidation, eutrophication, ozone layer depletion, acidification and marine ecotoxicity. These show the same variation over the value chain as GHG emissions.

The importance of catch does not imply, however, that all efforts should be concentrated on the fishing activity. In fishery a lot has been done in the past. The autoline equipment has automated previously manual operations thus increasing productivity and reducing product loss. Fishermen have learnt to be more careful when landing the catch. Often the fish hangs loose on the hooks, which means that they easily fall off when hauling the line. These fish are damaged and several studies show a high mortality. Hence fishermen try to bring these fish with a tool consisting of a long staff with a spike at the end. Ideally the spike should hit the head but if not used accurately it will hit the loins or other important part of the fish thus reducing both the economic value and the yield. Furthermore a great deal of effort has gone into developing new bait based on waste materials, such as gut and entrails. So far bait has been developed for other species such as haddock, but no successful cod bait has been developed.

Apart from changes in fishing practices the study identified a number of improvement options:

- Change of cooling agent to alternatives with less climate impact
- Reduction of cooling agent leakage.
- Use of electricity of guaranteed origin.
- Alternative methods for processing and packaging to reduce loss of product and increase the quality through the value chain.
- Reducing the proportion of frozen cod loins in favour of chilled loins to improve the economic value of the catch and hence reduce the environmental impact allotted to the other fractions.
- Better utilization of the fish, e.g. utilize the “earcut” fraction for products for human consumption instead of animal feed.
- Use bait made from waste materials instead of the current practise of using species that can be used for human consumption.
- Freeze counters in shops consume in general a lot of energy. Reduction measures include covering the counters during closing hours.
- This project has not investigated products from other catch methods (largely trawl, set nets and Danish seine). However a number of studies have from the North-eastern cod fishery and a number of related fisheries have clearly shown that autoline fisheries use less fossil energy and give a higher yield than the average fishery of this cod stock. <u>Hence a large environmental benefit could be realized if a larger proportion of the total cod catch would be taken by autoline equipment.</u>

The two first bullet options alone will give more than 30% reduction in climate gas emissions. This is shown in Fig. 4.

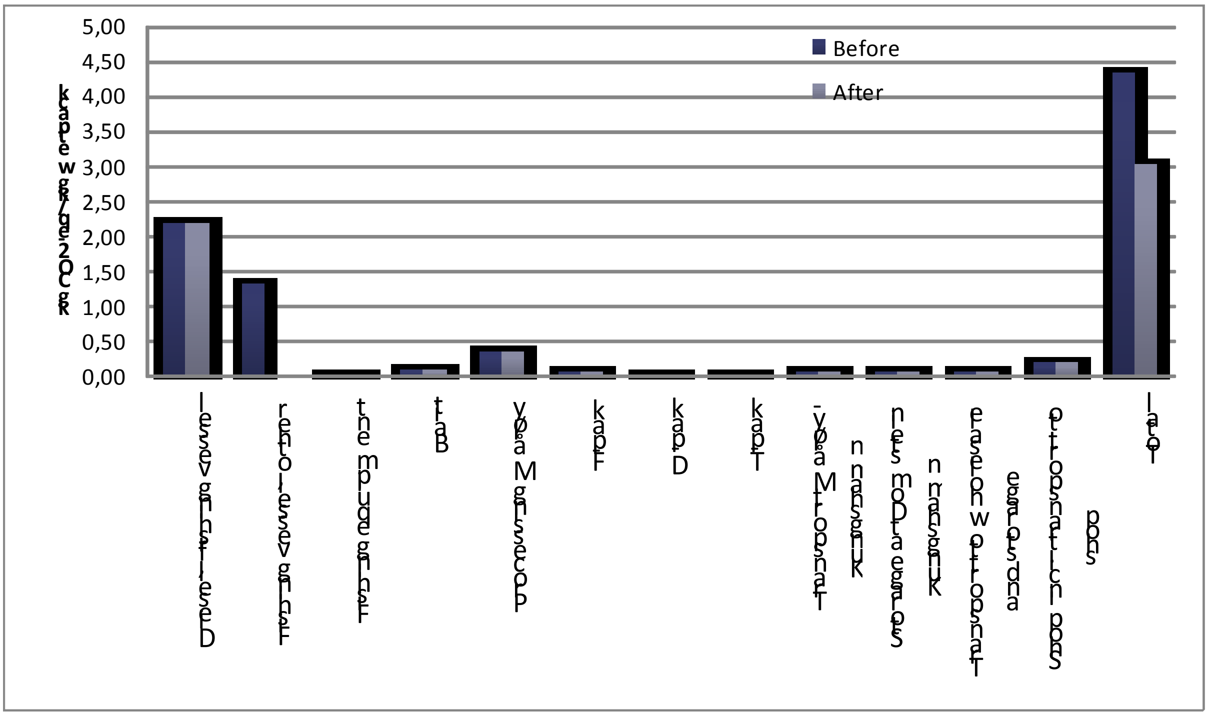

Fig. 4: Reduction of GHG emissions after introducing new cooling agent and reducing leakages.

Discussion

The results show that the earlier life stages are responsible for the major environmental impact of cod products. Using the analysis it was possible to identify the main contributors to environmental degradation. Through discussions between scientists and companies in the value chain, improvement options could be identified and assessed according to how easy it would be implement, technically and economically. The most important action taken was to replace the cooling agent onboard with an alternative with a much lower climate impact, and reducing cooling agent leakages. At a relatively low cost the climate impact was cut 30 %.

Other improvement actions have been carried out for a different reason. For example in order to comply with the KRAV requirements some measures had to be taken:

- Low sulphur fuel.
- Increased documentation activities.
- New tracking system installed on boats.

The preliminary results indicate that the switch to low sulphur fuel (0, 05 % S) has had a very limited impact on the environmental impact but there was no increased cost. This is partly due to the fact that the sulphur content of previously used fuel was already low (0.14 %) partly that NO_x emission dominates the acidification impact. The environmental benefit of the tracking system is difficult to assess but if it leads to an increase the market share of autolinecaught cod and reduce the share for e.g. trawled cod the net effect is an environmental benefit.

Because of the cost increase caused by the tracking system and the increased documentation requirements the price of ecolabelled wetpack is higher than non-ecolabelled wetpack but only 6 % higher.

Other improvement options are being considered now, e.g. utilizing fish better by reducing the waste proportion (gut and entrails) and increasing the yield for human consumption. In addition to the change of cooling system in the fishing boat and above mentioned measures that have already been taken the project will look at the environmental effect of a number of other measures. These measures also include measures that the companies in the value chain have no control over, e.g. reducing the energy consumption of freeze storage in shops.

The reason that the major impact of the other environmental effects (photochemical oxidation, eutrophication, ozone layer depletion, acidification and marine ecotoxicity) mainly comes from the fishing phase is probably due to two conditions:

1. Electricity production in the involved countries (Norway and Sweden) is largely sourced from hydropower and nuclear energy. Both these energy sources give low impact on climate change and the other categories.
2. The transport distances are relatively small. Most of the markets for Norwegian fish products are further afield.

Not all environmental effects of the fish products have been calculated. Biological effects, i.e. impact on ecosystems were not quantified. We assume that the impact on the seafloor is very low; in contrast with bottom trawling that scrapes large areas of seafloor while fishing. The impact on the cod stock is difficult but the autoline fisheries comply with strict regulations set up by the ecolabels KRAV and MSC.

The study has shown that methodological choices are of great importance for the end result. Especially the choice of allocation method is very important. The reason is partly the high level of bycatch and big differences in price between species, and partly the big difference in value between the different parts of the fish. This means that economic and mass allocation give very different results.

We have chosen economic allocation in both of the indicated parts of the analysis. Use of mass allocation was tested. The effect was a major decrease of environmental impact of the studied products while the animal feed has an increased environmental impact.

The main reason for using economic allocation was that this reflects the priorities of the fishermen and processers better than mass allocation. It also serves as a better basis for improvement analysis. Using this allocation encourages actors to take actions anyway makes sense to them because it is economically sound. Examples include

a) Increase the yield of products for human consumption
b) Reduce the level of bycatch of low economic value
c) Reduce loss and wastage all along the value chain

A big drawback with economical allocation is that prices are not constant. In fact they may vary a lot, depending on market conditions. This includes both the price of the cod and other species but also the economic value of the different parts of the cod. On the other hand the relative quantities of each species may also vary a lot. This will cause uncertainty of the results but for all allocation methods not only for economic allocation. Use of mass allocation would mean that the animal feed would “take” a lot of the environmental impact of the cod even though it has no economic value. Using mass allocation would mean that the environmental benefit of increase the yield for human consumption would be very small.

Conclusion

The study shows that cod products have a significant environmental impact. The study also clearly demonstrates that LCA has a large potential as a decision support tool. If the method is used in combination with economic tools and in open discussion with actors in the value chain the benefits are greatly enhanced. In our experience it is important to combine theoretical analyses and studies with inspections in the “field” to get a “hands-on approach”. Things are often different in the real world from what they are supposed to be.

Another lesson learned is that the LCA method, like any other analysis tool should be used with great caution. In this example we see that the allocation method chosen affect the results of the analyses very much. Hence great care must be taken before using the results for comparison with results from other analysis. One common application of food LCA results is to compare food products directly, for example by comparing climate impact of cod products with chicken products.

Future work:

Calculation of environmental effect of a number of improvement options in the autoline fisheries and further down the supply chain are still ongoing.

In the coming months research effort will be focused on coastal cod fisheries. Because of the small distances covered these vessels can supply fresh, high quality fish. However the small catches and large distances (from the fishing fields in Northern Norway to the main bulk of consumers in Southern Scandinavia and other markets further south) to consumers pose major challenges.

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Life Cycle Assessment of southern pink shrimp products from Senegal. An environmental comparison between artisanal fisheries in the Casamance region and a trawl fishery off Dakar including biological considerations

A. Emanuelsson[1, 2], A. Flysjö[1], M. Thrane[3], V. Ndiaye[4], J. L. Eichelsheim[5], F. Ziegler[1.]
1) SIK, Swedish Institute for food and Biotechnology
2) University of Gothenburg, Sweden
3) Aalborg University, Dep. of Development and Planning
4) CRODT, Centre de Recherches Oceanographiques Dakar-Thiaroye
5) IDÉE Casamance, NGO, Senegal
E-Mail: ae@sik.se

Keywords: Artisanal, developing country, discard, *Penaeus notialis*, Senegal, trawl.

Abstract

Life Cycle Assessment of two Senegalese seafood products exported to Europe was performed based on the functional unit (FU) of one kilogram of product (frozen whole shrimps) plus the accompanying package at the point of import to Europe. One product is produced by on-board processing demersal trawlers based in Dakar. The other production chain starts with the fishery in the Casamance river in southern Senegal were fishing is conducted by two different artisanal fisheries. Major differences between the three fisheries included (trawl, mujas and félé-félé) were shown using both classical environmental impact categories and extended biological ones, related to the FU (bycatch, discard, undersized target catch and seafloor disturbance). For the product originating from trawling, the fishing stage was the most important activity for all the investigated impact categories with high values for all biological categories except the undersized target catch. For the product originating from the artisanal fishery, processing and storage dominated most environmental impact categories, but with an overall lower impact load than the industrial trawl. However, high rates of smallsize target catch and lower but significant bycatches were documented for the artisanal fisheries. Finally, improvement options are discussed, and authors conclude that an increased traceability and labelling is desirable to make active consumer choices possible

Introduction

The main aim of the present study was to quantify the environmental impacts caused by a Senegalese shrimp product from fishing to market by performing a Life Cycle Assessment (LCA) of the artisanal fishery for southern pink shrimp (*Penaeus notialis*) in the Casamance region. Secondary aims were to compare the different fishing methods (artisanal and industrial) from an environmental point of view including utilized and non utilized bycatch (discards). Biological effects of the different fishing methods were included in the analysis and an additional goal was to attempt to quantify a few socio-economic indicators. This study was carried out as a collaboration between the Fisheries and Aquaculture Department at the Food and Agricultural Organization of the United Nations (FAO), the Swedish Board of Fisheries, the Swedish Institute for Food and Biotechnology (SIK), IDEE Casamance and Centre de Recherches Oceanographiques Dakar-Thiaroye (CRODT). The biological part of the study also resulted in a B.Sc. thesis where that part is presented in more detail Emanuelsson (2008).

The southern pink shrimp (*Penaeus notialis*) occurs in estuaries and coastal waters of West Africa from Mauretania to Angola, where it inhabits muddy sand bottoms at depths ranging from 2-100m.

The shrimp stock occurring in the Casamance estuary has its spawning grounds in the sea off the coast of Senegal and Guinea Bissau. After hatching and metamorphosis to various larval stages in the sea, juveniles migrate upstream in shallow areas of the river to feed and grow in the nutrient-rich mangrove areas that are found along the entire river. Three months later, adult shrimps migrate back to the sea in the central and deepest part of the river to spawn. (Lhomme 1984). While the fishery takes place all year round, landings have two peaks, with the largest in September-November after the rainy season in June to September, implying two salinity dependent cohorts (Matthews et al 2006; LeResete 1992)

There are mainly two artisanal fishing methods in use today:

Félé-félé. Drift nets used in intermediate parts of the river, around 120 m long and 1-2 m deep with 12 mm meshes (24 mm when stretched), trailed by canoes and actively managed by three men. *Mujas*. Stow net pairs of filtering trawl-like nets placed by one man on each side of an anchored canoe in the deepest part of the river during low tide, i.e. the fishery is powered by the tidal current that brings in the large shrimps migrating towards the sea.

The shrimp fishery in the Casamance is theoretically regulated by a system of fishing permits, by a minimum stretched mesh size of 24 mm and by a ban on pull nets and the capture, possession and trading of shrimps smaller than >200 individuals /kg. The Dakar-based fisheries are more large-scale. Vessels are diesel-driven and demersal trawls are used by the around 30 trawlers active in this fishery. The boats are out fishing for about 25 days. Fishing goes on all year, so a vessel can make around 10 mares a year. Most vessels are owned by foreign, European, companies. Reported landings in the Casamance varied between 800 and 1.200 tonnes between 2000 and 2006 (IDEE Casamance 2007). Total artisanal pink shrimp landings (including the Casamance region) represent on average 60% of total pink shrimp landings in Senegal which varied between 2.500 and 3.600 tonnes between 2004 and 2006. Consequently, around 40%, or 1.100-1.600 tonnes are fished in the trawl fishery described below (DPCA, Diarra Dioup unpubl.). No estimates of Cath-Per-Unit-Effort (CPUE) in the Casamance fisheries have been documented so far. However CPUE at sea, i.e. in the trawl fishery, decreased by over 90% between 1970 and 2005 (UNEP 2002, Samb et al 2007). Decreasing CPUE has also been documented recently for the five most commercially important species by the Senegalese oceanic research centre CRODT, which all can be found in the shrimp bycatch (Samb et al 2007).

Method / Approach

System boundary

The studied system starts with production of supply materials for the respective fisheries, e.g. fuel and gear material. Fishing is presumed to be undertaken by félé-félé and mujas nets 50% each with regard to total landings. In the case of the artisanal fishery, the shrimps are landed in the villages along the rivershore, where they are bought and transported by traders to the processing plants in Ziguinchor by a pick-up, cleaned and deep frozen before transport via warehouse to the port in Dakar. The study ends at the point of import, i.e. no further transport, storage, preparation or waste treatment is included, mainly due to the lack of data and the fact that the chains to be compared are identical from the point of export. The transport to Europe was included (even though it is the same in the two chains) as the role of long-distance food transports is often debated. In the case of the trawl fishery, processing, including packaging, is done at sea. The products are landed and taken for storage in Dakar where they are stored for, on average, 1-2 months. From there, the same type of transport on container freighters takes the product to the European market. The main market for shrimp product from trawl fisheries are Greece, Portugal and France.

Functional unit

The functional unit in the present study is ***one kilogram of frozen, whole, pink shrimps*** packed in a plastic bag inside a cardboard box, delivered to the port of Vigo, Spain. The shrimps originate either in the Dakar-based trawl fishery or in the Casamance artisanal fishery, assumed to be done by equal use of mujas and félé-félé nets with regard to total landings.

Allocations

In the fishing phase, several species are landed together and the allocation between them has been done on an economic basis. Especially in the trawl fishery, the amount of landed by-catch terms of weight is considerable (88%), while the economic importance of it is much less important (54%). Therefore, it is assumed that the shrimps are the driving force of this fishery rather than the fish that is also landed.

Data inventory

Data inventory of the foreground system in the Casamance was undertaken by local experts (IDEE Casamance and CRODT) in collaboration with the Swedish-Danish LCA team (SIK and Aalborg University) from November to December 2007. Relevant authorities and organisations were visited and existing documentation regarding the stock and the fishery gathered. Data for the Casamance fishery was collected by visiting fishing villages, interviewing fishermen and inspecting their catches upon landing.

Traders buying shrimps and taking them to the processing plants were also interviewed. Two processing plants in Ziguinchor were visited and technical staff answered questions with regard to production, logistics and the use of e.g. energy, refrigerants, packaging material, freshwater etc. Data for the background system, e.g. production of packaging materials, fuels and transports was taken from database Ecoinvent v.2.0. Electricity production in the Casamance was modelled based on information from the local producer.

In Dakar, the data inventory was undertaken in collaboration with a shrimp biology expert from Centre de Recherecheѕ Oceanographiques Dakar-Thiaroye (CRODT) from December 2007 to January 2008. With regard to the fishery, data from the two largest trawling companies was used. These two companies operate 15 and 4 shrimp trawlers, respectively and so 19 out of the total number of vessels of 30 were covered. Representatives of the companies provided data on landings, fuel use, use of refrigerants and logistics after landing. Information on the composition of different energy sources in average Senegalese electricity production (used in the present study for electricity use in the Dakar region) was found on the website of the International Energy Agency.

Data gathered of 30 landings in two fishing villages (around Ziguinchor and Bangangha, around 20 km upstream from Ziguinchor), constitute the basis for the artisanal biological part of the present analysis. Fishermen were either instructed beforehand to bring the entire catch ashore and sort it into landing and discard there or they were asked to estimate the weight and species discarded. Length distribution of landed shrimps was measured (carapace length) and landed by-catch was identified to species or genus and weighed, as were the landed shrimps. Local authorities' provided data for discard assessments onboard trawlers based on surveillance agreements with Mauretania and also records of total landings by species in terms of mass and economic value. The companies themselves provided length distribution data and boat inspections provided data for the seafloor disturbance model. Calculation setup was based on effective opening width with utter board length added, average speed, and average trawling time allocated economically to the yield per trip.

Method for Impact Assessment

The impact assessment method chosen here is CML 2001 (Guinée 2002) and the categories studied are Global Warming Potential (GWP), Acidification Potential (AP), Eutrophication Potential (EP), Photochemical Ozone Creation Potential (POCP), Ozone Depletion Potential (ODP), Human toxicity (hTox), Terrestrial toxicity (tTox), Marine Aquatic Toxicity (maTox) and Marine Sediment

Ecotoxicity (msTox) and Energy (E) as these categories were considered to be the most relevant ones for the chains studied. The category Global Warming Potential was updated with the new characterisation factors according to IPCC 2007. For energy the method Cumulative Energy Demand Pré (2007), in SimaPro, developed by Pré Consultants was used. The LCA was carried out in LCA software SimaPro v.7 (2007). In addition to the characterised LCA results, some biological aspects such as under sized individuals, bycatch, discard and seafloor impact are also displayed as biological impact parameters, by quantifying them and relating to the functional unit. Bycatch here is defined as all catch except target catch (*P.notialis*). Discard is defined as “the proportion of catch that is returned to the sea, in most case dead, dying or badly damaged” (Kelleher 2005), i.e the fraction of the bycatch which in not used.

Results

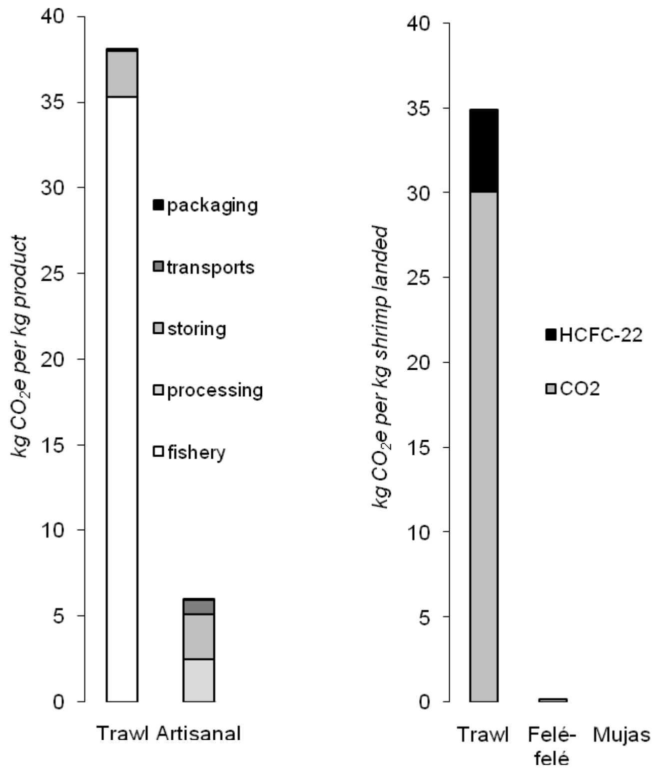

Fig. 1: Global Warming Potential caused by (left) a kilogram of shrimp product delivered to Vigo and fished either in artisanal fisheries or in the trawl (right) only the fishing stage contribution is divided into refrigerants leakage and combustion CO_2 . Note that artisanal HCFC leakage of HCFCs instead are included in processing phase.

As is evident from Fig. 1, the difference in Global Warming Potential (GWP) between artisanal and industrial fisheries is enormous due to the use of 9.8 l of diesel fuel and 2.7 g of refrigerant R22 in the trawl fishery as opposed to 0.05 l of fuel and no use of refrigerants in the fishing phase in the artisanal fisheries. It must be kept in mind, though that processing is included in the trawl fishery, which explains part of the difference. Over 35 kg of CO_2e are emitted per kg of shrimps landed in the trawl fishery, 0.2 kg in the félé-félé fishery and no global warming emissions at all in the mujas fishery. When the life-cycle after landing is added, the artisanal product causes emissions of 7.8 kg CO_2e per kg of product and the industrially fished product 38 kg CO_2e per kg. The major contributions to global warming emissions from the artisanal product are caused by energy- and refrigerant-related emissions in processing and storage.

The difference with regard to eutrophication is considerably larger and this category is dominated by emissions of nitrous oxides from combustion of fossil fuels in both chains. In the acidification category, the impact of the industrially fished product is three times higher than the artisanal one. The diesel fuel used in the trawl fishery has a sulphur content (0.4%) only 10 percent of the heavy fuel oil used for electricity production in the Casamance (4%), otherwise the difference would be even greater. The combustion and production of these fuels explain the main part of the acidification caused throughout the chains. Shipment also plays a role. As shown in figure 2 complemented by table 1, both comparing all impact categories between the two process lines - artisanal fisheries score 50- 60% lower in all toxicity categories with the exception of terrestrial toxicity which is higher for the artisanal product. This is due to the emission of mercury to soil from the batteries used. Many of the toxic emissions also originate from the production of fossil fuels. For the trawlers, the aquatic emissions of copper ions from the anti-fouling paint, accounts for a considerable part of the aquatic toxicity results.

The formation of ozone is largely correlated to the use of gasoline and to the production of fossil fuels: gasoline, diesel as well as heavy fuel oil. Gasoline in the chains studied only occurs in the félé-félé fishery stemming from the use of outboard engines. The diesel is used on the trawler and for transports and heavy fuel oil is used for electricity production. This is the category where transports score highest (almost 20% of the artisanal products emissions).

A refrigerant with a high ozone depletion potential, R22, is used both onboard the trawlers and in the processing plant on land. At the processing plant, two refrigerants are used, one for ice-making (R22), of which only the very low amount used for shrimps is allocated to the products and one for freezing and maintenance (R404a) which is entirely allocated to the shrimps. R22 has a high ODP and GWP, while R404a has zero ODP, but an even higher GWP compared with R22. Therefore R22 dominates this category while R404a is important in the category GWP.

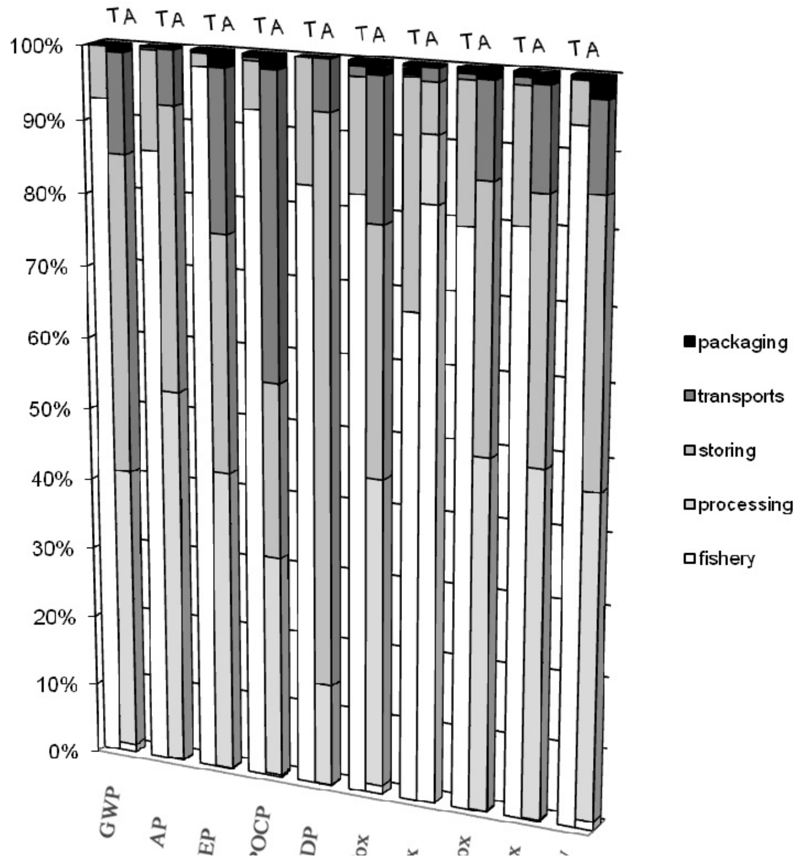

Fig. 2: Relative impact category contribution by product phase in trawl chain (T) and artisanal chain (A).

Tab. 1: Summary of results of comparisons between the three fisheries in the different environmental impact categories

Environmental Impact category	Félé-félé fishery	Mujas fishery	Trawl fishery	Data quality/ Uncertainty
Global warming	+ cooling agents, oil-based electricity	+ cooling agents, oil-based electricity	- fuel use in fishing, cooling agents, less oil-based electricity	good data on use of energy and cooling agents in processing and in trawl fishing, rather large uncertainty of fuel use in félé-félé fishery
Eutrophication	+ oil-based electricity	+ oil-based electricity	- NO_x from fuel use in fishing	good data on energy use in processing and fishing
Acidification	+ high sulphur fuel oil for electricity	+ high sulphur fuel oil for electricity	- less oil-based electricity, but high fuel use in fishing	good data on energy use in processing and fishing
Aquatic toxicity	+ no anti-fouling	+ no anti-fouling	- anti-fouling	high variation in fuel use data and estimations on emissions and content of copper
Terrestrial toxicity	- mercury batteries	- Mercury batteries	+ no mercury batteries	good data on battery use, estimations on mercury content and emissions

Overall better environmental performance is marked by (+), overall less good environmental performance is marked by (-). Main factors influencing this result (both positive and negative) is noted in text along with an estimate of data quality/variation/uncertainty.

Biological Impact Categories

A full stock assessment based impact category could not be included because of data deficiency, however size distribution, and relative yield size shows that both fisheries induces comparable amounts of extra mortality to the common stock measured in biomass, whilst the artisanal fishery catches mostly small pre mature individuals and the industrial mostly catch larger mature individuals. As an LCA biological parameter this can be described as less than 0.1 kg under sized shrimps per F.U. in the industrial case, compared with 1.4kg for Félé-félé and 0,4 kg for Mujas see Tab. 2.

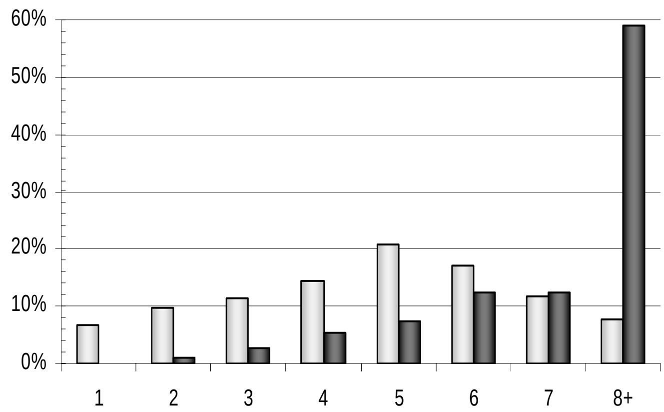

Fig. 3: Size distribution from 2005-2006 by a major company representing 60% of all industrial catches, compared with one out of three artisanal (black) distributor factories active in Casamance in Nov 2005. 1 is largest, 8+ is smallest legal and smaller.

Tab. 2: Summary of biological results.

Biological Impact Category	Artisanal		Industrial
	Félé-félé fishery	*Mujas fishery*	*Trawl Senegal*
Discard kg/kg FU	0.15	0.8	2.7
Bycatch kg/kg FU	0.25	1.2	7.3
Under sized kg/kg FU	1,4	0.4	0.09
Seafloor m^2/ kg FU	0	0	10100

All biological parameters are expressed as related to the functional unit.

By catches rates (88%) implies over 7 kg of non target catch per F.U. is caught in the industrial case, of which almost 3 kg (30%) are discarded back to the sea. Mujas, the worst artisanal gear catches around over 1.2 kg bycatch (54%) of which 0.8 kg (35%) are discarded. The FéleFéle rates are lower and thus corresponding mass per FU is 0.25 kg bycatch (25%) and 0.15 kg (15%) discard for every kg shrimps caught. Both artisanal fishing methods are approximated to null in their effective seafloor disturbance. Offshore trawlers however needs to sweep roughly one hectare for every kilogram of target catch.

Discussion

For all impact categories studied, the shrimps from the trawled fishery have a higher environmental burden, except for terrestrial toxicity, where artisanal fisheries have higher results because of the use of mercury-containing batteries. The main impact for the trawled shrimps is at the fishing stage, which also include processing and packaging. The use of fuel and refrigerants in the trawl fishery is very high and although there may be ways to decrease the fuel use onboard (Hassel *et al*. 2001), the type and amount of refrigerants used may be an easier improvement to achieve in the short-term.

Artisanal shrimps scored very low in term of resources used for fishing and the processing phase dominated the same categories as the trawling: energy, GWP and ODP. The source of energy used (and of course the amount) is very important for this result and an important improvement option would be to change from using average Casamance electricity to renewable energy sources. The use of refrigerants at the processing plant and storage was important from a global warming and ozone depletion perspective and a switch to less harmful refrigerants and/or decreased leakage represents important improvement option regarding in this respect. Looking at a future scenario, where the processing plant and ice production plants in the Casamance use solar energy for electricity production and an environmentally harmless refrigerant (NH_3), the global warming emissions of the artisanal product would decrease drastically to less than 4 kg of CO_2e/kg (half of today's emissions) and these would mainly be related to the storage in M'bour and transports. Whether or not this scenario is realistic is not judged here, but the example shows the potential of designing the chain on land of artisanal seafood products in an environmentally efficient way. Moreover, the use of mercury-free batteries and the collection of used batteries should be encouraged. Providing fishermen with environmentally friendly batteries could be an option.

On the biological side, stock assessment and relating fishing effort to its outcome is the basis of sustainable fishing practices. The use of a selectivity device, such as a species-selective grid, could be very favourable both in the trawl fishery and in the mujas fishery, decreasing the amount of discard and fish by-catch. That would decouple the fish fishery from the shrimp fishery and make it possible to optimise each of them. An increase in mesh size in both artisanal fisheries could also decrease the catches of undersized fish, something already suggested by the fishermen. The netting used today is of a "mosquito net type". Also, in artisanal fisheries, a spatial regulation could improve the catch composition of the félé-félé fishery. If it were conducted further upstream, a smaller proportion of small shrimps would be caught as the shrimps migrate upstream in the areas where félé-félé nets are set.

Conclusion

There are major differences between the artisanal fishery and the trawl fishery in all environmental impact categories included. Trawling uses much more fuel and refrigerants, and leads to considerably higher amounts of landed by-catch of fish, discard and seabed impact than the artisanal methods. Since processing is done onboard the trawlers, it is not completely fair to compare the fishing stage alone. The difference decreases when processing on land is added to artisanal fishing but still the trawl fishery leads to five times higher global warming emissions than artisanal fishing including processing. Transports and packaging only contribute a minor part to the overall result in both chains. The most important biological improvement options for the trawl fishery in addition to performing stock assessment and relating the fishing effort to its results, consist in implementation of more selective gears that separate the shrimp and fish catches from each other. Exchanging the refrigerants used onboard from so called synthetic (e.g. HCFCs and HFCs) to natural ones (e.g. NH_3 and CO_2) would result in considerable improvements in the categories ozone depletion potential and global warming potential.

Consumer pressure requires traceability and therefore traceability and labelling of the products as to origin in artisanal or industrial fisheries, perhaps even to distinguish between félé-félé and mujas fishery would be desirable to make active consumer choices possible. Intercontinental trade of seafood is sometimes debated as inefficient from a global warming emission perspective. The present study shows that frozen seafood products produced in developing countries in highly energy-efficient fisheries, like the studied artisanal fisheries, could well be environmentally competitive even on markets that are located far away from the fishery. Prerequisites are that the chain on land is designed in a resource-efficient way and that biological sustainability can be ensured

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Effect of structural and management characteristics on variability of dairy farm environmental impacts

M.S. Corson[1,2] and H.M.G. van der Werf[1,2]
[1] INRA, UMR1069, Soil Agro and hydroSystem, F-35000 Rennes, France,
michael.corson@rennes.inra.fr
[2]Agrocampus Rennes, UMR1069, Soil Agro and hydroSystem, F-35000 Rennes, France

Keywords: LCA, dairy farms, Brittany, organic, conventional, variability, environmental impacts

Abstract

Decision-makers in Brittany are seeking ways to reduce the environmental impacts of producing milk. We applied Life-Cycle Assessment (LCA) methodology (Guinée, 2002) to specialised dairy farms in Brittany to (1) estimate their potential environmental impacts, (2) characterise differences in impacts between conventional and organic production methods, and (3) identify production practices or factors with the greatest influence on impacts. We studied 46 conventional farms and 14 organic farms in Brittany. LCA calculations were performed with a tool called EDEN. For each farm, EDEN estimated farm-gate nutrient balances and potential direct (originating on the farm), indirect (originating prior to and off the farm), and total impacts for eutrophication, acidification, climate change, terrestrial toxicity, non-renewable energy use, and land occupation. Results showed significant differences in estimated potential impacts of organic and conventional dairy farms and largely agreed with previously published estimates of the effect of production mode on dairy-farm impacts. Variability of mean impact estimates by production mode was relatively small for acidification, climate change, and land occupation, but markedly larger for eutrophication and terrestrial toxicity. In the current study, we searched for factors to explain this variability by evaluating relations between and within input factors and impact estimates with standard statistical analyses. These results point toward farm characteristics that can influence particular environmental impacts of dairy farms the most, such as farm N balance for eutrophication impacts, manure imports for terrestrial toxicity, and feed imports for non-renewable energy use.

Introduction

In industrialised countries, nearly all farms that specialise in bovine milk production employ cattle bred for high milk-production rates. Most of them attempt to maximise (or at least optimise) this genetic potential by supplementing grass-based heifer diets with significant amounts of grain and concentrated feed. These feeds require nitrogen fertiliser and/or non-renewable energy to produce and process, thus increasing environmental impacts of these intensive systems. Other environmentally relevant emissions of dairy farms include ammonia, methane, and phosphorus emissions from manure, as well as methane production in bovine digestive tracts (Milne, 2005). In response to environmental consequences of agricultural activities, such as algal blooms and high nitrate concentrations in groundwater, decision makers in Brittany, France, are seeking ways to reduce environmental impacts of all farm types, including dairy farms.

To determine whether management changes influence a farm's environmental impacts, one first must be able to estimate these impacts quantitatively. Of the several methods developed to categorise and estimate the environmental impacts of agricultural production systems (Halberg *et al.*, 2005), Life Cycle Assessment (LCA) seems best suited because it can consider multiple impact categories and can do so as a function of both on- and off-farm activities (Guinée, 2002). We applied LCA methodology to dairy farms in Brittany to (1) estimate their potential environmental impacts, (2) characterise differences in impacts between conventional and organic production methods, and (3) identify production practices or factors with the greatest influence on impacts.

Method / Approach

We studied 60 specialised dairy farms from all 4 departments of Brittany: 46 conventional farms and 14 organic farms. The input data included information about the following farm characteristics over a one-year period: productivity and management of livestock, crops, and pasture; machinery; organic and inorganic fertiliser use; feed and forage use; pesticide use; energy-carrier and plastics consumption; and summary economic data (e.g., gross revenue). LCA calculations were performed with a Microsoft® Excel-based tool called EDEN (van der Werf *et al.*, submitted). For each farm, EDEN estimated farm-gate N, P, and K balances and used a modified CML2 characterisation method (Guinée, 2002) to estimate potential impacts for eutrophication (kg PO_4 equivalents), acidification (kg SO_2 equiv.), climate change (100-year horizon, kg CO_2 equiv.), terrestrial toxicity (kg 1,4-DCB equiv.), non-renewable (NR) energy use (GJ), and land occupation (m^2). EDEN distinguishes “direct” impacts that originate on the farm site itself from “indirect” (off-farm) impacts associated with the prior production and transport of supplies imported to the farm. The sum of direct and indirect impacts equals total impacts for a given impact category. Direct and total impacts were compared among farms by standardising them to two functional units: (a) 1 tonne of energy-corrected milk (ECM) sold and (b) on-farm plus estimated off-farm hectares utilised (van der Werf *et al.*, submitted). We used economic allocation (proportion of total gross revenue) to allocate impacts to each farm’s products (milk, animals, and crops).

The current study broadened statistical analysis of the dataset, first by examining correlations within and between sets of input factors and impact estimates. With regression analysis we then searched for reduced sets of factors to estimate indirect, direct, and total impacts within all impact categories but one (land occupation). We used personal knowledge, literature data, and input-impact correlations in the dataset to select input factors that have documented influence on the emissions used to calculate potential impacts in each category. Strong ($r>0.7$) correlations among some of the input factors allowed us to reduce the ensemble set among the 5 impact categories to 10 factors (units in kg unless noted): P imported in manure, N imported in fertiliser, N imported in feed, farm N balance, animal units (head), total uncorrected milk production, concentrated feed fed per dairy cow, diesel fuel used, mass of machinery owned, and usable on-farm agricultural area (ha).

We performed multiple linear regressions with these input factors to predict gross impact estimates, rather than impact estimates per functional unit, because the latter reflect values transformed by factors either moderately variable from year-to-year (i.e., proportion of income from milk sales, used for economic allocation) or themselves estimated by EDEN (i.e., off-farm hectares utilised to produce inputs). Additionally, adequate prediction of impacts calculated per a given functional unit suggests the use of input factors calculated per the same functional unit, which would have doubled the number of input factors required. We used and compared two criterion-based regression methods (Akaike’s Information Criterion (AIC) and Mallow’s *Cp* statistic) to find the best subset of predictors for each impact estimate (Faraway, 2002). The need for two models each for indirect, direct, and total impacts among 5 impact categories led to the calculation of 30 regression models.

Results

Potential impact estimates

When calculated per on- and off-farm ha, the mean estimates of total impacts calculated by EDEN showed significant differences between organic and conventional farms for all studied impacts, conventional farms having significantly greater impacts per ha (van der Werf *et al.*, submitted; Tab. 1). When calculated per tonne ECM, conventional farms had significantly greater acidification and terrestrial toxicity impacts, but significantly lower land occupation; total eutrophication and climate change impacts and NR energy use showed no significant differences per tonne ECM (Tab. 1). Estimated direct impacts followed the same patterns of significance as total impacts, with the addition of climate change impact, which was significantly greater on organic farms per tonne ECM (Tab. 1). Coefficients of variation of the mean estimates of total impacts by production method ranged from 13-28% for acidification, climate change, non-renewable energy use, and land occupation, 33-76% for eutrophication, and 93-238% for terrestrial toxicity (Tab. 1). Organic farms displayed greater coefficients of variation for mean total impact estimates than conventional farms. Coefficients of variation of direct impacts followed the same patterns as those for total impacts.

Tab. 1. Mean direct and total estimated impacts (and coefficients of variation) (1) per tonne energy-corrected milk (ECM) and (2) per ha of on-farm land (direct impacts) or on- and off-farm land (total impacts) occupied for organic and conventional farms (from van der Werf *et al.*, submitted). Symbols after group means indicate differences significant at (') p<0.05, (º) p<0.01, and (*) p<0.001.

Potential impact	Units	Location	per t ECM sold Organic	per t ECM sold Conventional		per ha Organic	per ha Conventional	
Eutrophication	kg PO_4 equiv.	Direct	4.8 (76%)	5.6 (41%)		23.1 (65%)	43.3 (38%)	*
		Total	5.1 (76%)	6.2 (39%)		23.2 (65%)	40.9 (34%)	º
Acidification	kg SO_2 equiv.	Direct	5.4 (16%)	4.8 (15%)	'	28.1 (18%)	37.3 (20%)	*
		Total	6.3 (20%)	7.2 (18%)	'	31.2 (18%)	48.3 (16%)	*
Climate change	kg CO_2 equiv.	Direct	910 (20%)	786 (18%)	'	4659 (15%)	6116 (21%)	*
		Total	1012 (20%)	942 (16%)		4960 (13%)	6321 (17%)	*
Terrestrial toxicity	kg 1.4-DCB equiv.	Direct	0.01 (208%)	1.48 (95%)	*	0.06 (208%)	11.05 (94%)	*
		Total	0.44 (252%)	1.67 (85%)	º	1.95 (240%)	10.81 (83%)	*
Non-renewable energy use	GJ	Direct	0.9 (28%)	0.8 (25%)		4.7 (33%)	6.1 (21%)	º
		Total	2.8 (26%)	2.9 (17%)		13.9 (27%)	19.2 (15%)	*
Land occupation	m^2	Direct	1969 (19%)	1315 (18%)	*	NA	NA	
		Total	2054 (19%)	1509 (14%)	*	NA	NA	

Correlation and regression analysis

Some input factors showed strong correlation, such as concentrate fed per dairy cow vs. milk produced per dairy cow (r=0.779) and N fertiliser imported per ha vs. pesticide active ingredients applied per ha (r=0.735). Certain total impact estimates also showed strong correlations, such as acidification vs. climate change (per ha, r=0.813; per tonne ECM, r=0.678) or acidification vs. eutrophication (per tonne ECM, r=0.633).

Tab. 2. For indirect, direct, and total estimated impacts in each impact category, coefficients of variation (R^2) of the best regression models selected by two criterion-based methods (Akaike's Information Criteria (AIC), Mallow's *Cp* statistic, or both) and the importance ranking (1=most important) of the factors selected in each model.

Impact	Location	Method	R^2	N imported in fertiliser (kg)	N imported in feed (kg)	Farm N balance (kg)	P imported in manure (kg)	Uncorrected milk production (kg)	Animal units	Diesel used (kg)	Machine mass (kg)	Concentrated feed fed (kg/cow)	Usable on-farm area (ha)
Eutrophication	indirect	both	0.894	2	1	4	6	5					3
	direct	both	0.999	3		1		4	5		6		2
	TOTAL	both	0.998		2	1			5		4		3
Acidification	indirect	both	0.881	1	2		3						
	direct	both	0.891			3		2	5	1		4	
	TOTAL	both	0.947	1	3		5	2	6	4			
Climate change	indirect	AIC	0.887	1	2						3	4	
		Cp	0.883	1	2						3		
	direct	both	0.865			1		2	4	3		5	
	TOTAL	both	0.922	2	3	5	7	4	1	6			
Terrestrial toxicity	indirect	AIC	0.166						3	2		1	
		Cp	0.118		1								
	direct	AIC	0.689	3		4	1		5	2			
		Cp	0.664				1			2			
	TOTAL	both	0.665				1			2			
Non-renewable energy use	indirect	both	0.844	1	2				4		3		
	direct	both	0.957					2		1		3	
	TOTAL	both	0.884	2	1				5	3	4		

Regression models selected by the AIC and *Cp* methods included 1-7 input factors (mean = 4.4) and had coefficients of determination (R^2) ranging from 0.664-0.999, except for indirect terrestrial toxicity (R^2=0.118-0.166) (Tab. 2). As expected, models with better R^2 values tended to include more factors (r=0.630). Models for indirect impacts tended to contain fewer factors (mean=3.7) than those for direct (mean=4.5) or total (mean=5.0) impacts. Both methods of model selection chose the same model for all impact estimates except indirect climate change (AIC selected one additional factor), indirect terrestrial toxicity (no common factors), and direct terrestrial toxicity (AIC selected 3 additional factors) (Tab. 2). Within impact categories, regression models usually predicted direct impacts best (highest R^2 values in 3 of 5), followed by total impacts (2 of 5) (Tab. 2). Regression models of indirect impacts tended to have the lowest R^2 values (4 of 5) (Tab. 2). Considering the three models for each impact category, mean R^2 values were ranked in this order: eutrophication, acidification, NR energy use, climate change, and terrestrial toxicity; the ranking for total impacts followed the same pattern except for a change in rank between NR energy use and climate change (Tab. 2).

By impact category and model-selection method, only 3 of 10 sets of indirect- and direct-impact models shared common input factors: eutrophication (4 of 6 factors shared for AIC and *Cp* methods) and terrestrial toxicity (2 of 3-5 factors shared for the AIC method) (Tab. 2). For each factor except usable on-farm area, inclusion in indirect- and direct-impact models showed a skewed distribution, with inclusion in one set of models (e.g., indirect impacts) at least twice as frequent as inclusion in the other set (e.g., direct impacts). Factors that appeared more often in indirect-impact models included N imported in feed, N imported in fertiliser, P imported in manure, and mass of machines owned (Tab. 2). Factors that appeared more often in direct-impact models included diesel fuel used, milk production, concentrated feed fed, farm N balance, and animal units (Tab. 2). When selected, N imported in fertiliser and N imported in feed tended to rank among the two most important factors predicting impacts among all categories (Tab. 2).

Direct vs. indirect impacts

For both production modes, EDEN estimated that the majority (65-96%) of total potential impacts occurred as direct impacts on the farm site itself, except for NR energy use and, for organic farms alone, terrestrial toxicity (Tab. 3). Between production modes, a greater percentage of total impacts occurred on organic farms than on conventional farms, except for terrestrial toxicity.

Tab. 3. By impact category, percentage of total estimated potential impacts occurring as direct impacts on organic and conventional farms.

Potential impact	Organic	Conventional
Eutrophication	95	90
Acidification	87	65
Climate change	89	81
Terrestrial toxicity	8	85
Non-renewable energy use	34	27
Land occupation	96	85

Discussion

Potential impact estimates

Total impact estimates made with EDEN largely agreed with previously published estimates of the effect of production mode on dairy-farm impacts (Cederberg and Flysjö, 2004; Basset-Mens *et al.*, 2009; Thomassen *et al.*, 2008). These studies found that organic farms, characterised by lower inputs and larger surface areas, tend to have lower impacts per ha than conventional farms; however, their consequently lower milk production pushes their impacts closer to those of conventional farms when impacts are expressed per unit of milk produced. The small sample size of organic farms (n=14) seems the most likely explanation for the larger variability observed in organic-farm impact estimates. Some variability in impacts was undoubtedly due to rounded estimates (e.g., of feeds bought and crops sold) and missing values among the survey data.

Correlation and regression analysis

Strong correlations observed between certain input factors were consistent with known characteristics of dairy farms (e.g., feeding more concentrate increases milk production, farms with more surface area use more diesel fuel, greater N fertiliser use presages greater pesticide use). Strong correlations observed between certain impact estimates reflected the fact that some impact categories aggregate the same emissions (NH_3 to air for eutrophication and acidification) and that some processes have emissions classified into multiple impact categories (diesel use emits both CO_2 (climate change) and SO_2 (acidification)).

In 3 of 15 cases, the *Cp* method selected more parsimonious models than the AIC method, which was expected (Faraway, 2002), but not so infrequently. Because both methods selected similar models, except for one notable exception, either method seems appropriate for model selection. The exception was for indirect terrestrial toxicity, for which the *Cp* model was entirely different from the AIC model. The *Cp* method did consider the model selected by AIC, but rejected it for having too many factors to be worth the relatively small increase in *Cp* value. The change in ranking of R^2 values for total impacts (compared to that for direct impacts) between NR energy use and climate change may have reflected the fact that most NR energy use was indirect while most climate change impacts were direct. Most models of indirect impacts shared no factors in common with models of direct impacts, highlighting the ability of these regression models to deduce EDEN's method for separating indirect and direct impacts.

Direct vs. indirect impacts

The predominance of direct impacts over indirect impacts in most impact categories emphasises the major contribution of on-farm processes to potential environmental impacts. Indirect impacts dominated NR energy use, primarily the energetic costs of producing energy carriers, machines, concentrated feed, and inorganic fertilisers. Theoretically, farmers have greater control over direct impacts than indirect impacts. Although reducing fuel use, feed imports, and machinery purchases can decrease the magnitude of indirect impacts, farmers personally have little influence on the prior impacts of and resources used for a given unit of input. The greater predominance of direct impacts on organic farms vs. conventional farms reflected the relatively lower (or zero) imports of inorganic fertilisers, feeds, and pesticides by organic farms. In contrast, since organic farms did not import manure (the main source of heavy metals) and exported most heavy metals in milk and animals, direct impacts for terrestrial toxicity were negligible, leading to the domination of indirect impacts due to imports of feed and forage (though still relatively small in absolute terms).

Effect of structural and management characteristics on impact variability

Even the lowest variability observed among potential impacts per functional unit, 13-21% for acidification and climate change impacts, indicate that management practices have room for improvement on farms with above-average impacts for a given production mode. The more important factors selected for regression models indicate farm characteristics or management activities that could receive greater attention for reducing a given impact or its within-group variability, such as farm N balance for eutrophication impacts, manure imports for terrestrial toxicity, or feed imports for NR energy use.

Conclusion

Total impact estimates made with EDEN largely agreed with previously published estimates of the effect of production mode on dairy-farm environmental impacts per ha and per unit of milk produced. Considerable variability in estimations existed, however, with coefficients of variation of total impacts ranging from 13-28% for acidification, climate change, non-renewable energy use, and land occupation, 33-76% for eutrophication, and 93-238% for terrestrial toxicity. Both methods of regression-model selection (Akaike's Information Criteria and Mallow's *Cp* statistic) selected similar models and predicted most indirect, direct, and total impacts with R^2 values ranging from 0.664-0.999. The majority of total potential environmental impacts occurred due to on-farm activities (i.e., direct impacts) for eutrophication, acidification, climate change, and land occupation impacts; only for non-renewable energy use did off-farm activities predominate. Although the relatively small sample size needs to be increased, these results begin to indicate which management changes could reduce particular environmental impacts of dairy farms the most. Previous studies have not separated total impacts into direct and indirect components; it would be informative for future studies to do so.

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Life-cycle energy and greenhouse gas analysis of a large-scale vertically integrated organic dairy in the U.S.

M. Heller[1], S. Cashman, K. Dick, D. Przybylo, W. Walter, G. Keoleian
Center for Sustainable Systems, School for Natural Resources and Environment

University of Michigan, Ann Arbor, MI, USA

[1] mcheller@umich.edu

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Abstract

Organic agriculture has sustained consistent growth in the U.S. over the past decade, but very little systemic environmental impact benchmarking has been performed. This study is the first life cycle assessment (LCA) of a large-scale, vertically integrated organic dairy in the U.S. Data collected at Aurora Organic Dairy farms and processing facilities were used to build a LCA model for benchmarking the greenhouse gas (GHG) emissions and energy consumption across the entire milk production system, from organic feed production to transport of packaged milk. Overall GHG emissions were 1.7 kg CO_2eq per liter of packaged liquid milk. The major GHG contributors include enteric fermentation (27% of total) and feed production (22% of total). The energy consumption for the entire system was 15.7 MJ per liter of packaged liquid milk. Potential strategies for reducing the system GHG emissions are discussed.

Introduction

Agriculture is responsible for nearly seven percent of the total U.S. greenhouse gas emissions; over half of this is from livestock (USDA, 2004). The U.S. organic food sector has consistently grown between 15-20% annually over the past decade. Organic dairy in particular has grown by upwards of 25% in recent years (OTA, 2007). While such growth is in general lauded as an environmental success, there is a great need for systemic benchmarking of the environmental impact of organic agriculture in the U.S. in order to provide guidance for continual improvements in the sustainability of this rapidly growing sector.

Life cycle assessment (LCA) is a method for integral analysis of the environmental impact of products, processes or services by including all phases of the life cycle. Originally developed for the evaluation of industrial products and processes, LCA has proven a useful tool for evaluating complex agricultural systems such as dairy production (de Boer, 2003). LCA methodology has been used to compare the environmental performance of conventional and organic milk production in Sweden (Cederberg & Mattsson, 2000), Germany (Hass *et al*, 2001), Finland (Grönroos *et al*, 2006), and the Netherlands (Thomassen *et al*, 2008); and to assess the greenhouse gas (GHG) emissions from milk production in Ireland (Casey & Holden, 2005). The entire milk supply chain (farm production, transport, milk processing, packaging) in Spain (Hospido *et al*, 2003) and Sweden (Sonesson & Berlin, 2003) has also been analyzed with LCA methods.

This report describes method and model development, as well as energy use and GHG emission results, for a LCA of a large-scale, vertically integrated organic dairy in the U.S. Aurora Organic Dairy (AOD) is a leading U.S. provider of private-label organic milk and butter, managing over 12,000 milking cows and processing over 84 million liters (22 million gallons) of milk annually. Milk from six AOD farms (three in Colorado and three in Texas) is processed in a state-of-the-art processing facility in Colorado and then distributed to retail outlets across the country. Recent growth and a commitment to sustainability and the organic industry have led AOD to evaluate its life cycle GHG emissions and explore reduction strategies. This effort represents the first comprehensive LCA of large-scale milk production in the U.S., as well as the first LCA of a vertically integrated organic dairy.

Methods

Objectives

The purpose of this study is to conduct a LCA for a large, vertically integrated organic dairy in the US. The objectives of this research were the following:

- To highlight processes which contribute the greatest energy and GHG impacts across the overall system.
- To use the total energy consumption and GHG emissions as a benchmark for improvement.
- To identify and evaluate possible strategies for GHG and energy reduction within AOD's organic dairy system.

Functional unit

The functional unit (FU) for the entire milk production system is 1 liter of packaged liquid milk transported to distribution centers. "Packaged liquid milk" represents a mix of AOD's products ranging from skim to whole milk. Results were also analyzed based on energy corrected milk (ECM) at the farm gate in order to draw comparisons to existing studies. ECM considers the fat and protein content of the raw milk. ECM is calculated according to Bernard (1997), using the following equation:

$$\text{ECM (kg)} = 0.3246\cdot(\text{kg}_{\text{milk}}) + 12.86\cdot(\text{kg}_{\text{fat}}) + 7.04\cdot(\text{kg}_{\text{protein}}) \qquad (1)$$

System boundary

The processes investigated in this study are shown in Fig. 1. The time frame for the analysis is one year (April 2007-March 2008). The LCA starts with the production of feed on supplier farms and ends with the delivery of packaged milk to distribution centers across the U.S.; it includes all activities at AOD's six dairy farms and their company milk processing plant. Transport of animal feed and all milk products are accounted for in this study. Production of butter and powdered milk, both AOD products processed at co-packing facilities, are not included in this study due to insufficient data. Upstream burdens associated with farm milk and cream that is processed into butter and powdered milk are allocated away from the liquid milk system. The life cycle under investigation ends with delivery to the distribution centers and does not include transport to or refrigerated storage in retail outlets or consumer homes. Major building materials for farm and processing plant buildings are included and amortized over 50 years for farm buildings and 30 years for milk plant buildings. For completion, estimates of employee transport as well as corporate office operation are also included.

System description

Organic milk is produced on six AOD-owned dairy farms (three in Central and Eastern Colorado, two in Central Texas and one in the Texas Panhandle). The farms primarily purchase high quality organic alfalfa (as well as silages when available) for roughage fodder. All farms purchase the same organic grain pre-mix consisting of, on average, 40% corn, 10% barley, 12% wheat midds, 21% soybean meal, and 5% minerals. The overall diet for a typical mature dairy cow is 42% alfalfa, 42% grain pre-mix, and 16% other grass hays and silages. Feed supplier farms are generally located in the Western U.S. All cows are given access to pasture; therefore, pasture intake represents a portion of the cow's overall feed energy, but not a majority of it.

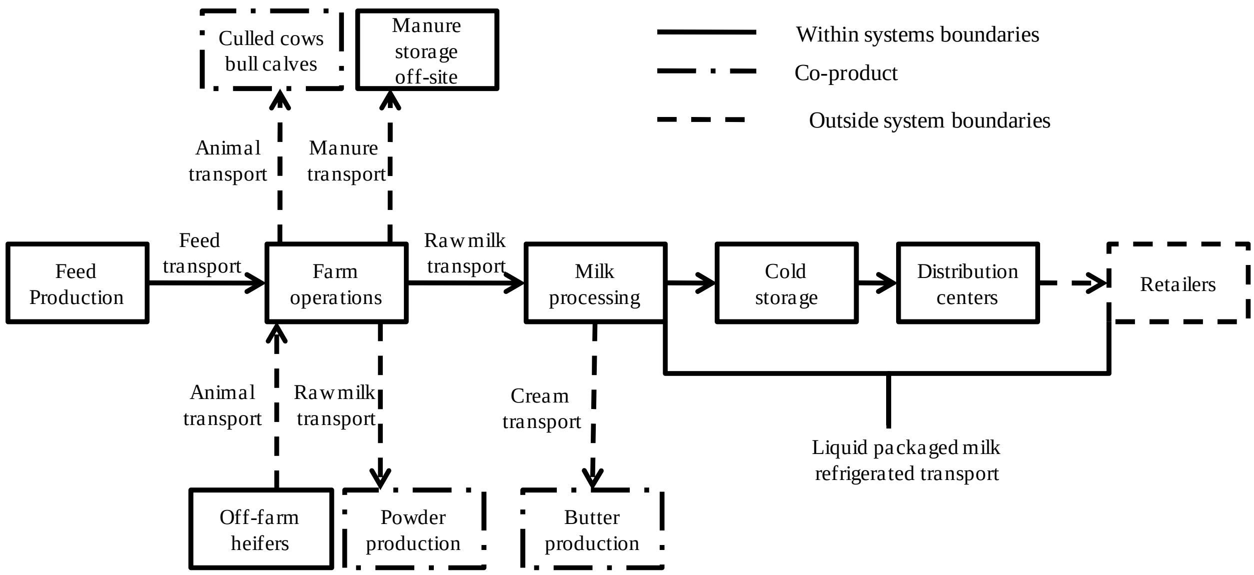

Fig. 1: Process flow for organic milk production. Arrows represent physical movement of materials and boxes represent different phases of milk production. Processes accounted for in this study are shown in solid lines while processes not accounted for are shown in dashed lines. Similarly, any material transport that is included in the study is shown as a solid arrow while material transport not included is shown as a dashed arrow. Processes (co-products) with upstream burdens allocated away from the liquid milk system are shown in dashed-dot lines.

All raw milk processed at AOD's ultra-pasteurization milk plant, located in Central Colorado, originates from AOD farms. During the time frame for this analysis, AOD produced 84 million liters (22 million gallons) of liquid milk. Final liquid milk is packaged in two styles: by the gallon and by the half gallon. Half gallon packaging dominates the product line, accounting for 98% of all final liquid milk packaging types. Half gallons are packaged in a gable top carton constructed of plastic coated paperboard. Gallon packaging is manufactured at the AOD milk plant using high density polyethylene (HDPE) in a blow molding process. All final liquid packaged milk is stored in corrugated cardboard boxes, wrapped in low density polyethylene (LDPE) film, and shipped on wooden pallets. All liquid milk products are shipped first to a nearby cold storage site, and then distributed throughout the U.S. via refrigerated tractor-trailer trucks.

Life-cycle assessment model and data collection

A model was created to calculate the GHG emissions and energy usage associated with the production of one liter of packaged, delivered milk. The assessment model was constructed using the LCA software, SimaPro 7.1.6, in accordance with the ISO 14040 LCA standards (ISO, 1997).

A large portion of the model data was collected onsite at AOD's farms and processing plant. These primary data include amount of feed, electricity, fuels, and packaging used over the one year time frame. Transportation distances for the shipment of feed, raw milk, and final packaged milk were collected from AOD records. Life cycle GHG emissions and energy consumption from production of fuels, building materials, dairy supplies, and packaging materials were calculated using databases available through SimaPro. Feed production was modeled with Ecoinvent version 2.0 datasets available in SimaPro (Ecoinvent, 2007). Fuel consumption and related GHG emissions from transportation were modeled using average US tractor-trailer datasets from Franklin Associates, 1998. Refrigerated transport was estimated to consume an additional 1.89 liters of diesel per hour of operation (Franklin, 2008), with only final packaged products refrigerated during transport. Product cold storage was estimated to consume 82 kWh per 10,000 liters of storage (Franklin, 2008). Regional electricity grids were modeled according to Kim & Dale (2005).

GHG emissions due to enteric fermentation, manure management, and industrial wastewater treatment were estimated according to chapters 10, 11, and 6, respectively, of the 2006 IPCC *Guidelines for National Greenhouse Gas Inventories*. Global warming potentials were characterized using the IPCC 2001 methodology using a 100-year time horizon (23 and 296 for methane and nitrous oxide, respectively) (IPCC, 2001). Energy resource impact was assessed using a modified Eco-indicator 95 (version 2.04) characterization: renewable energy flows (biomass, solar, wind, geothermal) were not included in the sum. Energy flows are reported on a LHV basis.

Co-product Allocation

Multiple economic outputs or co-products are common in agricultural systems. While system expansion is recommended to avoid co-product allocation (ISO, 1998), it is often not possible or practical for agricultural systems. In this study, co-product feedstuffs (e.g., soybean meal, a co-product in the production of soybean oil) were allocated on a mass basis. Additional allocation methods are described below.

Bull calf and culled cow allocation

In previous studies, allocation between meat (bull calf and cull cow) and milk co-products has been based on economics (Hospido *et al*, 2003; Thomassen *et al*, 2008; Grönroos *et al*, 2006), mass (Grönroos, *et al*, 2006) or energetics (Casey and Holden, 2005; Cederberg and Mattsson, 2000). Here, we use a causal relationship based on the energy (in the form of feed) needed to produce the meat co-product. Bull calves are sold shortly after birth on AOD farms. All energy and GHG burdens resulting from the production of the calf (i.e., pregnancy energy requirements) are therefore subtracted from the liquid milk system. Pregnancy energy requirements were calculated using equations from *Nutrient Requirements of Dairy Cattle 2001* (NRC, 2001). The amount of feed required to supply pregnancy energy was determined based on a typical cow diet. The calculated energy usage and GHG emissions from this feed production were then subtracted from the liquid milk system.

In the case of end-of-life culled cows, the total body mass present at the time of removal is allocated to the culled cow. Assuming that the cow's empty body mass is 18.8% fat and 16.8% protein, the energy of the body mass was determined using the following energy densities: 39.3 MJ/kg of fat and 23.4 MJ/kg of protein (NRC, 2001). The body mass energy was converted into an equivalent amount of feed based on a typical cow diet. The calculated energy usage and GHG emissions from this feed production were then allocated to the culled cows (subtracted from the fluid milk system).

Cream and milk powder allocation

Impacts associated with raw milk used for producing milk powder are allocated away from the liquid milk life cycle system on a milk solids basis as described by Feitz *et al*. (2007). Similarly, burdens associated with excess cream shipped from the milk plant to a butter co-packer are allocated away from the fluid milk life cycle on a milk solids basis.

Results

Base model results

Model results on a functional unit basis are shown in Tab. 1. For raw milk at the farm gate, 1.01 kg CO_2eq were emitted and 7.01 MJ of energy were consumed per kg of ECM. Over the full liquid milk life cycle, 1.71 kg CO_2eq were emitted per liter of packaged liquid milk, and the full life cycle energy consumption was 15.69 MJ/liter.

Tab. 1: GHG emissions and energy consumption per volume of packaged liquid milk transported to distribution centers.

	GHG	energy
per liter	1.7 kg CO_2eq	15.7 MJ
per gallon	6.5 kg CO_2eq	59.4 MJ

Life cycle distribution of GHG emissions

GHG emissions by individual processes in the milk production system are shown in Fig. 2. Methane produced during enteric fermentation contributes the greatest emissions on a CO_2 equivalents basis, accounting for 27% of total system GHGs. Organic feed production is the next largest contributor, making up 22% of total GHG emissions, with feed transport contributing 8% to total GHG emissions. Manure management accounts for only 6% of total emissions. The other large GHG contributor to the system is final product storage and transport, which accounts for 12% of total emissions.

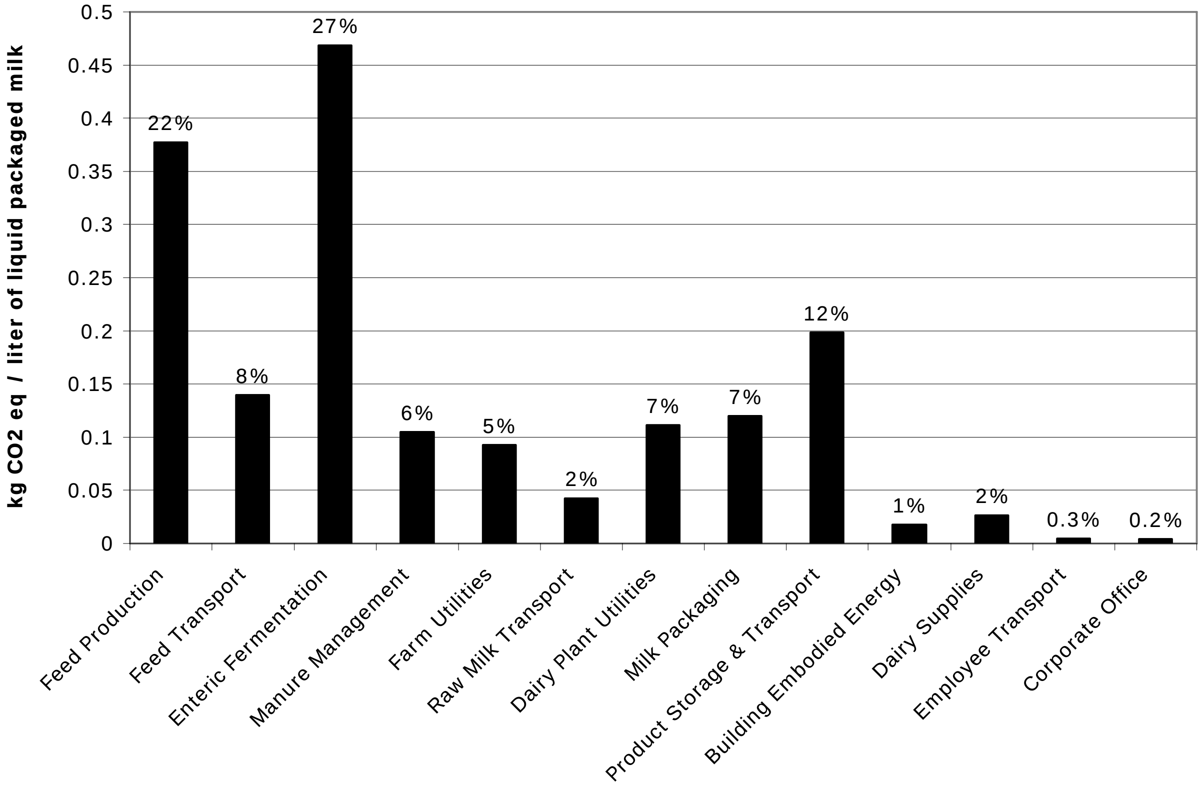

Fig. 2: Distribution of life cycle GHG emissions for packaged liquid milk.

Life cycle distribution of energy consumption

Energy use by individual processes in the milk production system is shown in Fig. 3. Feed production is the largest energy input, accounting for 25% of all energy usage. Transportation of feed from supplier farms to AOD farms accounts for 12% of the total energy consumption. Farm utilities make up 12% of total energy usage, whereas dairy processing plant utilities account for 14% of total energy usage. Other important energy contributors include product packaging, which makes up 9% of total energy usage, and final product storage and transport, which accounts for 18% of total energy usage.

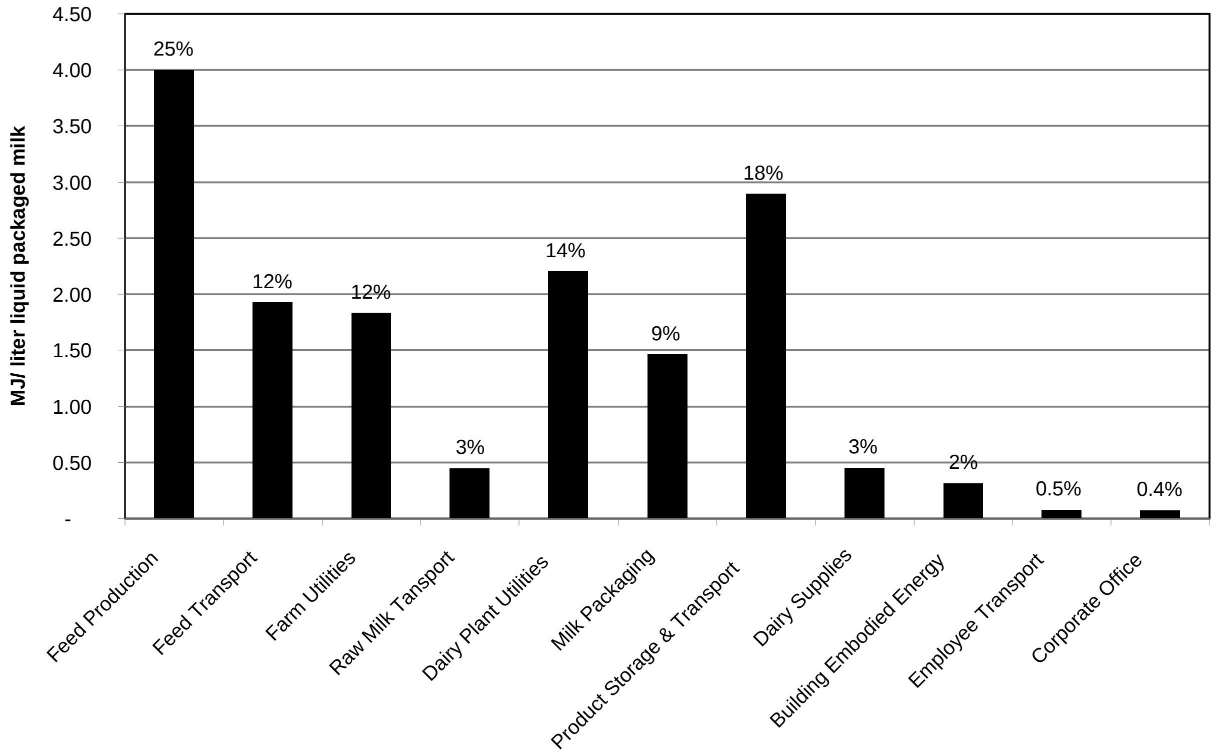

Fig. 3: Distribution of life cycle energy consumption for packaged liquid milk.

Discussion

Literature comparisons

Caution must be exercised in comparing life cycle results from this study with other results published in the literature. Differences in methods and model parameters can influence the comparison and lead to inaccurate conclusions. Because LCA is still under development and each country has its own agricultural practices and climate there are significant differences in the development and application of LCA even among European countries. Tab. 2 shows the range of values from reported LCA studies of milk production and processing. Results of this study fall within the range of reported GHG emission values, but outside the range of reported energy consumption values. There are many possible explanations for the discrepancy in energy values. AOD farms rely heavily on energy-intensive concentrated feeds that are often transported large distances. Given the scale and national distribution of AOD operations, there is also significant transportation of raw milk and final packaged milk. Indeed, transportation of feed and milk account for 30% of the overall system energy consumption. Energy of transport alone, however, does not account for the discrepancy with literature values. This study also includes contributions typically excluded in other studies, such as building embodied energy, employee transportation, corporate office activities, and a detailed account of purchased items. These secondary components, however, tend to make negligible contributions to the overall energy demand. Unfortunately, most dairy LCA studies in the literature do not report detailed stage-level contributions, so tracing the discrepancies is difficult. It is likely due to a combination of factors: higher energy demand for transportation and feed production (see below section), combined with lower (relative to literature studies) methane contributions to GHG emissions due to the high digestability of concentrated feed.

Feed Production datasets

This analysis relied on available LCA datasets for feed production. No LCA datasets exist for U.S. organic feed production of feed types purchased by AOD. LCA datasets, specifically for agriculture, are more prevalent for European conditions than for the U.S. U.S. conventional datasets were only available for corn, soybeans, and soybean meal. The base model considered in this analysis uses U.S. conventional datasets for corn, soybeans, and soybean meal, and Swiss (CH) organic datasets for all other feed types. The base model feed datasets were chosen to represent first geographic accuracy and second farming practices. To explore the effect of this assumption, two other feed scenarios were considered: all CH organic datasets and all CH conventional datasets. Overall, there is about a 6% increase in feed energy values and a 22% increase in feed GHG values when utilizing all CH conventional datasets rather than the base model datasets (Tab. 3).

Tab. 2: Comparison of literature reported LCA studies of milk production and processing.

at farm gate (per kg ECM)				
GHG (kg CO_2eq/kg ECM)	energy (MJ/kg ECM)	country	**C**onventional or **O**rganic?	reference
1.1	-	US	**C**	Phetteplace (2001)[a]
0.81	1.4	Spain	C	Hospido (2003)
1.3 - 1.5	-	Ireland	C	Casey (2005)
1.0	3.6	Sweden	C	Cederberg (2000)
0.95	2.5	Sweden	O	Cederberg (2000)
0.89	3.7	Netherlands	C	de Boer (2003)
0.92	3.9	Netherlands	O	de Boer (2003)
1.3	2.7	Germany	C, intensive	Haas (2001)
1.3	1.2	Germany	O	Haas (2001)
1.4	5.0	Netherlands	C	Thomassen (2008)
1.5	3.1	Netherlands	O	Thomassen (2008)
-	5.3	Finland	C	Grönroos (2006)
-	2.8	Finland	O	Grönroos (2006)
1.0	7.0	US	O	this study
total life cycle (per liter)				
(kg CO_2eq/ liter packaged milk)	MJ/ liter packaged milk			
1.05	6.2	Spain	C	Hospido (2003)[b]
-	6.4	Finland	C	Grönroos (2006)
-	4.4	Finland	O	Grönroos (2006)
1.7	15.7	US	O	this study

[a]not reported as a LCA study
[b]does not include delivery of final packaged milk

Tab. 3: Feed GHG and energy values when using different LCA datasets. Both absolute values and percent changes are displayed for energy and GHG.

	base model	CH Organic dataset	CH Conventional dataset
feed production (MJ)	359,000	337,000	373,000
percentage difference	X	-4.0%	6.4%
feed production (kg CO_2eq)	34,800	35,000	42,400
percentage difference	X	0.70%	22 %

Abatement options

Four abatement strategies were considered for this analysis: animal husbandry techniques, anaerobic digestion, biodiesel use for on-farm equipment, and on-farm wind power. The largest contributor to overall GHG emissions for all processes considered was enteric fermentation. Enteric fermentation emissions can potentially be reduced through certain animal husbandry techniques such as changing feed type (Monteny *et al*, 2006). For example, Grainger *et al* (2008) observed a 21% (milk solids basis) reduction in CH_4 production with the addition of whole cottonseed to the cow's diet. Such techniques are preliminary and require additional research, as well as market development of organic sources of promising alternative feedstuffs. Anaerobic digesters offer a means of reducing GHG emissions by capturing CH_4 produced during manure handling/storage and flaring the captured CH_4 for energy utilization. While installing an anaerobic digester with electricity generating capacity would reduce GHG emissions by offsetting grid electricity, the potential for reducing CH_4 emissions from manure management is limited because AOD's primarily solid-based manure management system already has low CH_4 emissions relative to other dairy farms using liquid-based manure management. Biodiesel is readily available in the region and can easily be substituted in most diesel engines at least at a 20% rate. A substitution of 20% biodiesel for farm diesel could potentially reduce overall GHG emissions by 0.2%. All AOD farms are located in regions of high wind potential, with Colorado ranking 11th and Texas ranking 2nd in the U.S. (AWEA, 2008). Displacing conventional grid electricity on all of the AOD farms offers a potential 2.5% reduction in overall GHG emissions.

Conclusion

The overall life cycle GHG emissions from a large-scale vertically integrated organic dairy in the U.S. were found to be 1.7 kg CO_2eq per liter (6.5 kg CO_2eq per gallon) of liquid packaged milk. Enteric fermentation was the most GHG intensive process, contributing 27% of GHG emissions to the total system. Energy usage was found to be 15.7 MJ per liter (59.4 MJ per gallon) of liquid packaged milk. Livestock systems are a significant emitter of GHGs in the U.S. Further LCA studies should be conducted in the U.S. dairy industry to understand the impact of alternative practices, and to allow more accurate national comparisons. In particular, life cycle models of organic feed production in the U.S. are needed. This life cycle assessment and other comprehensive studies provide important metrics to guide the dairy industry towards enhancing its overall environmental sustainability.

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Meat and milk products in Europe: Impacts and improvements

B. Weidema[1], J. Hermansen[2] and P. Eder[3]
[1] 2.-0 LCA consultants, www.lca-net.com, bow@lca-net.com
[2] Faculty of Agricultural Sciences, Dept. of Agroecology, University of Aarhus, www.agrsci.dk
[3] European Commission, JRC, Institute for Prospective Technological Studies, www.jrc.es

Keywords: hybrid LCA, monetarisation, beef, rebound effects, socio-economic assessment, cost-benefit, discounting, consequential LCA

Abstract

The overall environmental impacts from consumption of meat and dairy products in EU-27 have been assessed by the use of hybrid life cycle assessment (input-output data supplemented by specific process data). For the impact assessment, we applied a flexible model that allows results to be presented both in 15 traditional environmental midpoint indicators (global warming potentials, photochemical ozone creation potential, etc.) and in monetary units (Euro). Specifically for this project, a damage model for aquatic eutrophication was developed. We identified and quantified the improvement options for all processes contributing more than 10% to each of the midpoint impact categories. Rebound effects, synergies and dysergies of the different options were taken into account and we show the importance of rebound effects and interrelationships of the improvement options, as well as market constraints. The environmental impacts were monetarised and a separate socio-economic assessment performed, thus allowing a cost-benefit assessment of the improvements. We also analysed the significance of discounting. Uncertainties and limitations of the study are discussed.

Introduction

A study entitled "Environmental improvement potential of meat and dairy products" has been performed as a scientific contribution to the European Commission's Integrated Product Policy framework, which seeks to minimise the environmental degradation caused throughout the life cycle of products. A previous study (the EIPRO study) had shown that food and drink is responsible for 20% to 30% of the environmental impact of private consumption in the EU, with meat and dairy products contributing most. This study first presents a systematic overview of the life cycle of meat and dairy products and their environmental impacts, covering the full food chain. It then provides a comprehensive analysis of the improvement options that allow reducing the environmental impacts throughout the life cycle. Finally, the report assesses the different options regarding their feasibility as well as their potential environmental and socio-economic benefits and costs.

Inventory analysis method

The methodology applied in this study is a hybrid life cycle assessment method, which implies a system model that combines the completeness of 'top-down' input-output matrices, based on national accounting statistics combined with national emission statistics (known as NAMEA matrices), with the detailed modelling of 'bottom-up' processes from process-based life cycle assessments. Among the processes included in the model there are 15 agricultural processes (including different livestock production systems as well as feed production systems), 20 food and feed industry sectors, four household processes (such as food storage and cooking) and seven waste management processes. These specific processes are embedded into a newly developed NAMEA matrix for EU-27. The data for the agricultural processes are derived from detailed production models, including all relevant inputs and outputs. For example, for each of the five dairy farming systems the production model includes the specifications for different types of land use, herd composition, input of different types of feed, production output (milk, beef, cereal surplus), fertiliser application and nitrogen balance. Well-documented biological input-output relations, such as nutrient balances, have been used to specify the agricultural production models. Data on production volume, area, number of livestock by Faostat have been used to scale the production models up to the level of EU-27.

The functional unit of the study is the entire annual consumption of meat and dairy products in the EU-27. The reference flows include all meat and dairy products, except eggs and fats, all related restaurant and other catering services, shopping activities, storage, cooking and dishwashing in the households, tableware and household utensils, and waste treatment of food and packaging.

Impact assessment method

The impact assessment method used is Stepwise2006 version 2.1, a flexible model building on IMPACT2002+ v. 2.1 and the EDIP2003 methods, allowing results to be presented both in 15 traditional environmental midpoint indicators (global warming potentials, photochemical ozone creation potential, etc.) and in monetary units (Euro). The impact assessment method is reproduced as annexes to the report (Weidema *et al*. 2008).

Results for EU-27 meat and milk products

We find that the consumption of meat and dairy products contributes 24% of the monetarised environmental impacts from the total final consumption in EU-27, while constituting only 6% of the economic value. The contributions for each of the 15 midpoint impact categories are given in Fig. 1.

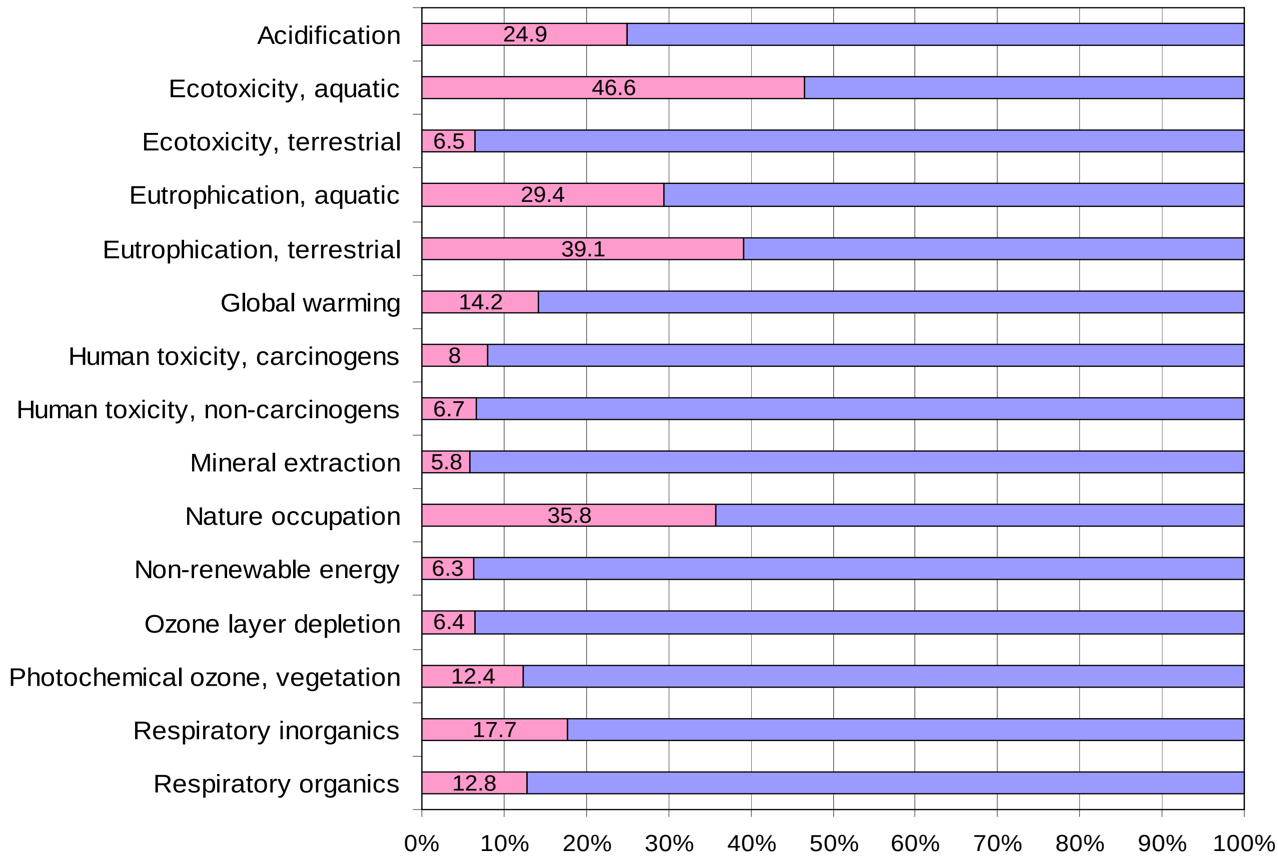

Fig. 1: Percentage contribution of meat and dairy products to the environmental impacts of EU-27 total final consumption.

For the impact categories that contribute the most to the overall monetarised impacts, the contribution of meat and dairy products varies from 14% (for global warming) to 36% (for nature occupation) of the impacts from the total final consumption in EU-27. The monetarised environmental impacts (externalities) are of considerable size compared to the private costs of the products (from 34% of the private costs for pork to 112% of the private costs for beef), see Tab. 1.

Tab. 1: Impact per EUR consumption expenditure for the four main product groups. Note that consumption expenditure includes all life cycle costs, i.e. also costs for shopping and meal preparation, and thus more than just the price of the products.

Impact category	**Unit**	**Dairy products**	**Beef**	**Pork**	**Poultry**
Midpoint categories:					
Acidification	m^2 UES	0.21	0.37	0.15	0.30
Ecotoxicity, aquatic	kg-eq. TEG water	305	298	389	252
Ecotoxicity, terrestrial	kg-eq. TEG soil	1.4	1.6	1.2	1.6
Eutrophication, aquatic	kg NO_3-eq.	0.021	0.028	0.016	0.023
Eutrophication, terrestrial	m^2 UES	0.83	1.60	0.57	1.27
Global warming	kg CO_2-eq.	1.65	2.47	1.07	1.12
Human toxicity, carcinogens	kg C_2H_3Cl-eq.	0.0030	0.0053	0.0022	0.0043
Human toxicity, non-carcinogens	kg C_2H_3Cl-eq.	0.0026	0.0032	0.0022	0.0031
Mineral extraction	MJ extra	0.013	0.013	0.011	0.013
Nature occupation	m^2 arable land	1.94	5.06	1.16	2.93
Non-renewable energy	MJ primary	20	24	18	20
Ozone layer depletion	kg CFC-11-eq.	4.5E-07	6.0E-07	3.4E-07	5.7E-07
Photochemical ozone, vegetation	m^2*ppm*hours	15	25	12	11
Respiratory inorganics	kg PM2.5-eq.	0.0018	0.0036	0.0012	0.0027
Respiratory organics	person*ppm*hours	0.0017	0.0027	0.0012	0.0012
Endpoint (damage) categories:					
Impact on ecosystems	Species-weighted m^2*years	2.8	6.1	1.8	3.4
Impacts on human well-being	QALY	1.3E-06	2.6E-06	8.9E-07	1.9E-06
Impacts on resource productivity	EUR	0.034	0.064	0.023	0.046
All impacts	EUR	0.53	1.12	0.34	0.67

The four main product groups (dairy, beef, pork and poultry products) contribute respectively 33-41%, 16-39%, 19-44%, and 5-10% to the impact of meat and dairy products consumption in EU-27 on the different environmental impact categories, see Fig. 2.

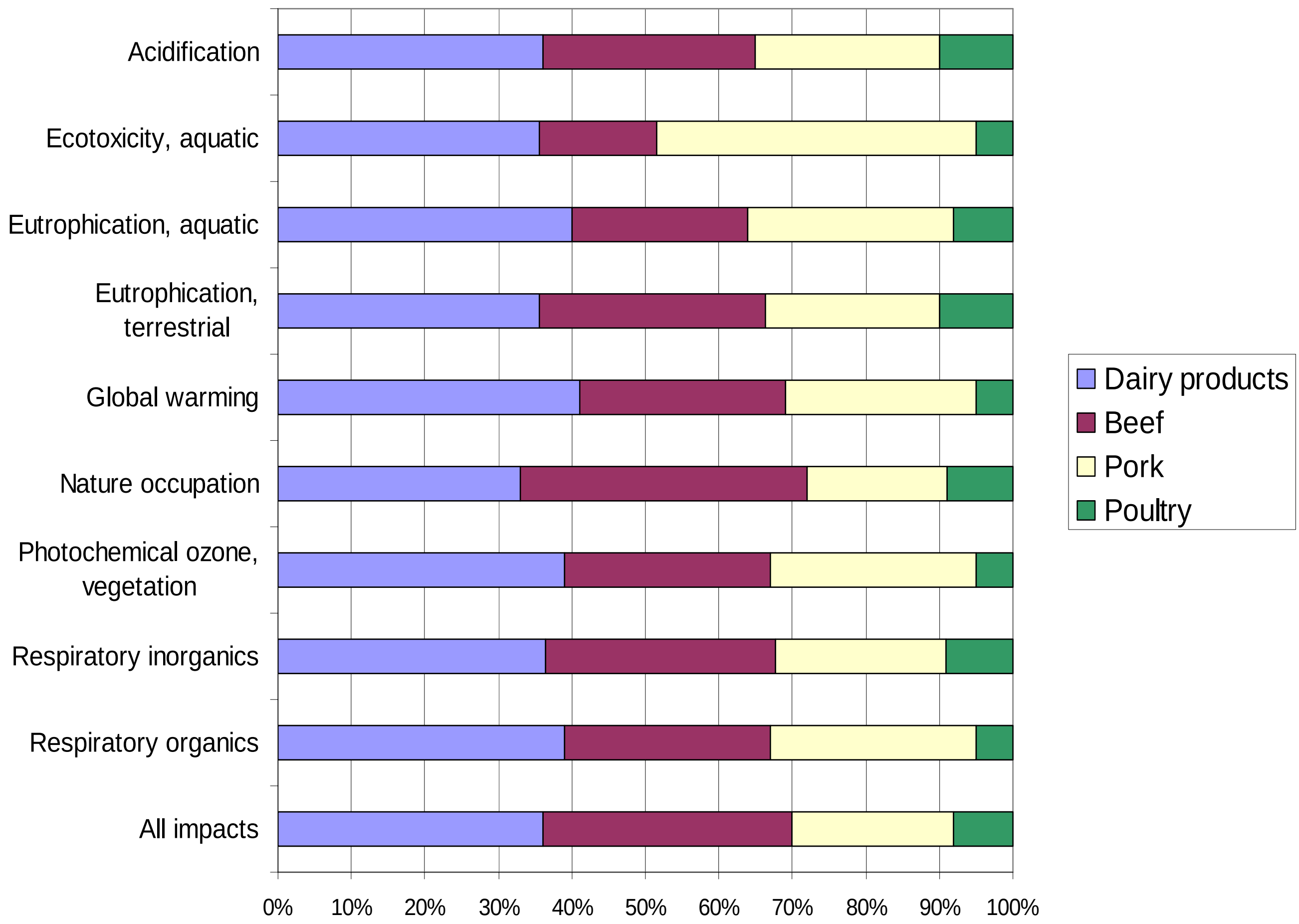

Fig. 2: Relative contribution (%) of the four main product groups.

Per kg slaughtered weight, there is a clear difference between the three types of meat, with beef having 4 to 8 times larger environmental impacts than poultry and up to 5 times larger than pork, see Tab. 2. These differences are less pronounced when comparing the environmental impact intensity (impact per Euro spent) of the three types of meat, see Tab. 1, where pork generally has the lowest impact intensity (down to 40% of the impact of poultry and 23% of the impact of beef), with the exception of aquatic ecotoxicity where pork production contribute with high copper emissions.

The values in Tab. 2 are significantly larger than for previous LCA studies on meat and dairy products. The inclusion of wholesale, retail and household processes causes an increase of 10% in the values compared to the values at the gate of the food industry. The remaining difference (up to 300% of previous studies) can be ascribed to the larger completeness of the hybrid life cycle assessment method.

Tab. 2: Impact per weight unit for the four main product groups.

Impact category	Unit	Dairy products	Beef	Pork	Poultry
		per kg raw milk equivalent	per kg slaughtered weight	per kg slaughtered weight	per kg slaughtered weight
Midpoint categories:					
Acidification	m^2 UES	0.30	4.32	1.55	0.98
Ecotoxicity, aquatic	kg-eq. TEG water	447	3471	4073	815
Ecotoxicity, terrestrial	kg-eq. TEG soil	2.1	18.9	12.8	5.2
Eutrophication, aquatic	kg NO_3-eq.	0.031	0.325	0.164	0.075
Eutrophication, terrestrial	m^2 UES	1.2	18.6	6.0	4.1
Global warming	kg CO_2-eq.	2.4	28.7	11.2	3.6
Human toxicity, carcinogens	kg C_2H_3Cl-eq.	0.004	0.062	0.023	0.014
Human toxicity, non-carcinogens	kg C_2H_3Cl-eq.	0.004	0.037	0.023	0.010
Mineral extraction	MJ extra	0.018	0.153	0.117	0.042
Nature occupation	m^2 arable land	2.8	58.9	12.2	9.5
Non-renewable energy	MJ primary	30	276	193	65
Ozone layer depletion	kg CFC-11-eq.	6.5E-07	7.0E-06	3.5E-06	1.8E-06
Photochemical ozone, vegetation	m^2*ppm*hours	23	288	121	37
Respiratory inorganics	kg PM2.5-eq.	0.0027	0.0417	0.0127	0.0086
Respiratory organics	person*ppm*hours	0.0025	0.0318	0.0129	0.0038
Endpoint (damage) categories:		0	0	0	0
Impact on ecosystems	Species-weighted m^2*years	4.1	71	18	11
Impacts on human well-being	QALY	2.0E-06	3.0E-05	9.3E-06	6.2E-06
Impacts on resource productivity	EUR	0.05	0.75	0.24	0.15
All impacts	EUR	0.77	13.00	3.52	2.16

Improvement options

We identified and quantified the improvement options for all processes contributing more than 10% to each of the midpoint impact categories.

More specifically 12 improvement options studied were:

1. Planting catch crops during winter (to reduce nitrate leaching, saving artificial fertiliser and the corresponding N_2O and ammonia emissions);
2. Improved growing practise and intensification of cereal production where yields are low today;
3. Optimised protein feeding in pig and dairy farming (to reduce NH_4 emissions and N leaching);
4. Liquid manure pH reduction (to reduce ammonia emissions);
5. Tightening the rules of manure application (to reduce nitrate leaching and N_2O emissions);
6. Copper reduction in dairy cattle and pig diets (to reduce copper emissions);
7. Methane-reducing diets for dairy cattle (to reduce methane emissions);
8. Biogasification of manure from dairy cows and pigs (to reduce methane and N_2O emissions);

9. Home delivery of groceries (to reduce air emissions related to car driving);
10. New cold appliances only A+ or A++ (to reduce electricity consumption);
11. Power saving in farming, food industry, retail, and catering;
12. Household meal planning tools (to reduce food losses and thereby all environmental interventions throughout the life cycle).

When all the identified environmental improvement potentials are taken together, the total improvement amounts to a reduction of 17 % for nature occupation, around 25 % for global warming and respiratory inorganics, 31 % for acidification and terrestrial eutrophication, 43 % for aquatic eutrophication, to 68 % for aquatic ecotoxicity (when rebound effects and synergies have been accounted for). Since the first three impact categories make up 95 % of the aggregated (monetarised) environmental impact, the aggregated improvement potential amounts only to about 20 % of the total environmental impact of meat and dairy products in EU-27 (and significantly less if rebound effects were not accounted for). Fig. 3 shows how much the environmental impacts may be reduced for the main environmental impact categories.

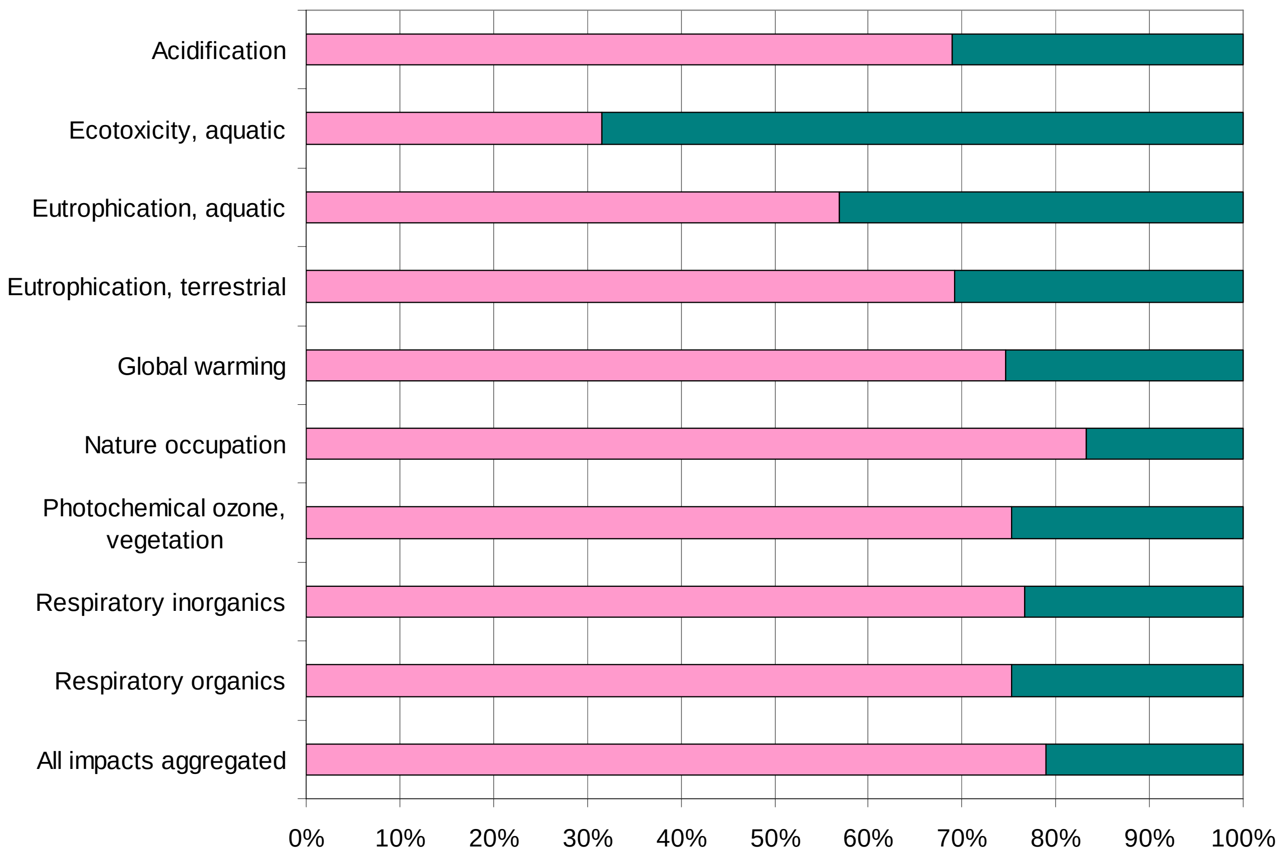

Fig. 3: Remaining and avoided environmental impacts of meat and dairy products if all identified improvement options are implemented together. Rebound effects as well as synergies and dysergies between different improvement options are considered.

Rebound effects, interrelationships and market constraints

We have quantified three types of rebound effects:

- Price rebound effects (environmental effects of the reduced or increased consumer spending when the improvement is more or less costly than the current technology),

- Traffic rebound effects (environmental effects of increased car driving when road congestion is reduced due to less car driving for shopping), and
- Technology rebound effects (environmental effects of the wider implementation of the improvement options than just for meat and dairy products consumed in the EU-27, i.e. also for exported products, and for food production and consumption activities in general).

The importance of the rebound effects is illustrated in Tab. 3.

Tab. 3: Rebound effects of 12 improvement options for meat and dairy products in EU-27. All values in MEUR per year. Negative values signify an improvement (= cost reduction).

ID no.	Economic impacts (costs)	Net environ mental impacts	Result before rebound effects	Rebound effects	in % of result before rebounds
1	70	-140	-70	-270	390%
2	-500	-2600	-3100	-4700	150%
3	1360	-3200	-1840	-1430	78%
4	900	-3500	-2600	-1260	49%
5	-590	-1620	-2200	-115	5%
6	210	-510	-300	-220	73%
7	0	-1280	-1280	-225	18%
8	1360	-2430	-1070	-1100	102%
9	-78000	-900	-78900	-7760	10%
10	-330	-320	-650	-370	57%
11	-620	-1100	-1720	130	-7%
12	-15000	-5300	-20300	640	-3%

Also interrelationships (synergies and dysergies) of the improvement options play an important role. While rebound effects increased the overall improvement potential from 9.3% to 16%, expected synergies add another 5% to the improvement potential, bringing it up to 21% of the total impacts. The main expected synergy is between home delivery of groceries and the adoption of meal planning tools.

The use of consequential modelling (taking into account market constraints) for the improvement options was only important for two of the improvement options namely biogasification of manure and cold appliances regulation, where the consequential modelling showed improvements of 128% and 187% of the results with the attributional modelling that does not take into account market constraints. In both cases the main reason for the difference was the larger emissions of the unconstrained electricity supply.

Socio-economic assessment and overall results

A separate socio-economic assessment was performed, in which the following issues were assessed:

- Direct production costs / Consumer expenditure
- Injuries
- Dietary health (mainly important for meal planning tools, but very uncertain)
- Supply security (only qualitatively described)
- Well-being of animals in human care (only qualitatively described)

- Landscape maintenance (only qualitatively described)
- Employment (assessed to be insignificant)
- Household work (reduced time usage for shopping)
- Income distribution (mainly important for home delivery of groceries, but not included in final assessment)

thus allowing an overall cost-benefit assessment of the improvements as shown in Fig. 4.

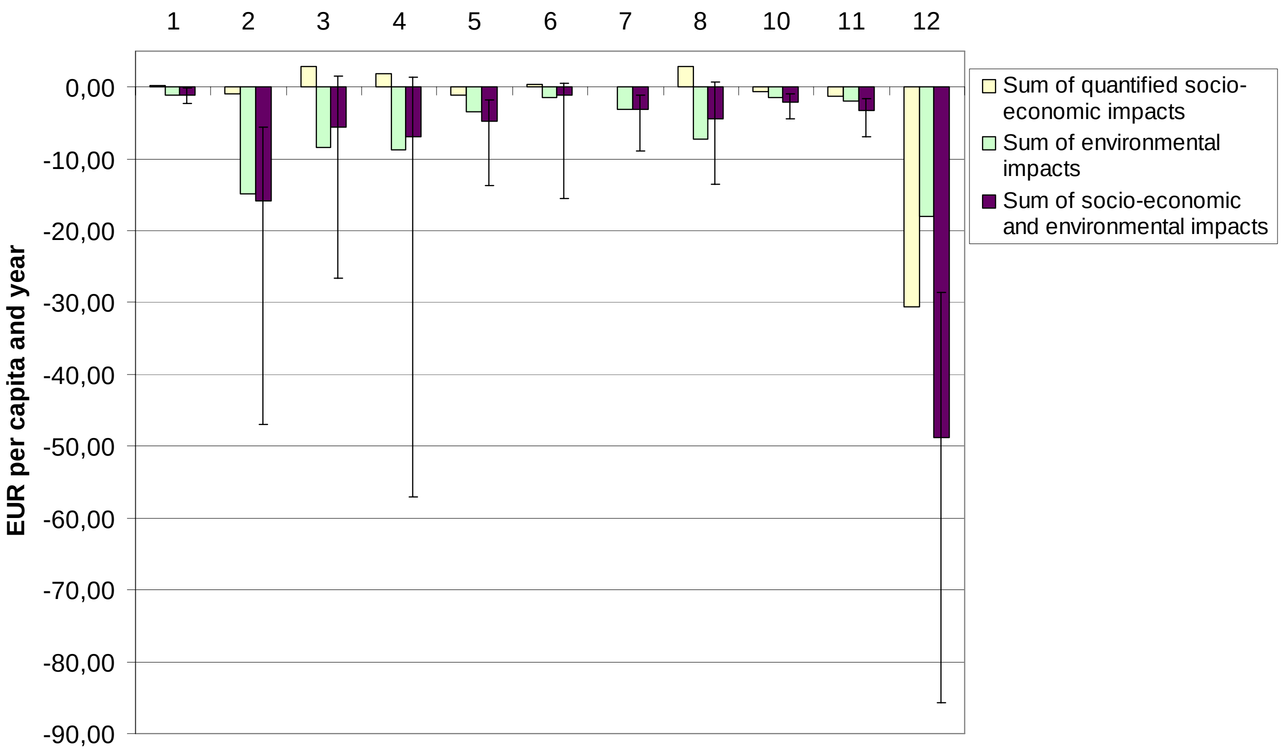

Fig. 4: Overall socio-economic and environmental impacts of improvement options in EUR per capita per year. The numbers 1 to 12 refer to the improvement options as given in Tab. 4.

Discounting

We analysed the significance of discounting at a 3% constant annual discount rate, with investments in capital equipment placed at year 1 (and therefore not discounted), changes in operating costs and related emissions distributed equally over the lifetime of the capital equipment and discounted accordingly, and a very simplified assumption for reductions in environmental impacts, namely an equal distribution over 100 years for global warming and nature occupation, and over 10 years for all other impacts.

Tab. 4 shows the net present value and the internal rate of return for the 12 improvement options. The internal rate of return is undefined for options where both direct costs and environmental impacts show a benefit. It should be noted that the internal rate of return *cannot* be used to prioritise between improvement options. For this purpose, the net present value is the most appropriate.

Tab. 4: Undiscounted annual net benefits, net present value under 3% constant annual discount rate, and the internal rate of return for the 12 improvement options

Improvement option	Annual net benefits, undiscounted	Annual net benefits, net present value	Internal rate of return
	MEUR	**MEUR**	**%**
1. Catch crops	521	224	47.00%
2. Cereal intensification	7790	4760	undefined
3. Optimised protein feeding	2770	970	12.10%
4. Liquid manure pH reduction	3350	2630	76.60%
5. Tightening of manure regulation	2310	1360	undefined
6. Copper reduction in animal diets	520	420	48.80%
7. Methane-reducing animal diets	1510	520	undefined
8. Manure gasification	2160	-40	2.80%
9. Home delivery of groceries	95400	83500	undefined
10. New cold appliances only A++	1020	350	9.10%
11. Power saving in industry	1580	1050	undefined
12. Household meal planning	23900	18300	undefined

Uncertainties and limitations

The main uncertainties are:

- For the majority of the improvement options, the overall uncertainty on the environmental improvement is dominated by the assumption of the degree to which the improvement option can be implemented, i.e. the area for which catch crops can be implemented, the actual cereal yields that can be achieved, the level of reduction in emissions, the extent of the power saving, and the extent that household behaviour can be affected. For the uncertainty on the aggregated impacts shown in Fig. 4, the uncertainty on the characterisation factors is dominating.
- For improvement options involving large changes in direct production costs, the uncertainty on the cost estimates may contribute significantly to the overall uncertainty. This is particularly the case for cold appliances regulation and for biogasification of liquid manure; see also Tab. 4.
- For some improvement options, the uncertainty on the socio-economic impacts dominates the overall uncertainty. This is particularly the case for home delivery of groceries (large, but very uncertain savings in household time usage) and meal planning tools (possibly large, but very uncertain impacts on dietary health, not included in the presented quantitative results).

Most improvement options show a net benefit at the 95% confidence level, but due to the large uncertainties in the characterisation factors, this is not the case for the four agricultural improvement options with the largest direct economic costs: Optimised protein feeding, liquid manure pH reduction, copper reduction in dairy cattle and pig diets, and liquid manure biogasification. This also makes these improvement options more sensitive to temporal discounting; see Tab. 4. Particularly the benefit of copper reduction is uncertain, since it depends on the impact potential of metal emissions, which may be overestimated in current characterisation models.

A number of impacts have been entirely omitted from the study (impacts from occupation of extensive grazing lands, disruption of archaeological heritage, antibiotic resistance, species dispersal, noise, pesticides transmitted through treated food, depletion of phosphate mineral resources), some have been modelled only very coarsely (all area uses treated equally, despite large differences in biological value) and some have been only qualitatively touched upon (erosion and water balance). Likewise, a number of rebound effects, synergies/dysergies, and socio-economic impacts have not been quantified, but only described qualitatively. It is likely that these short-comings mainly bias the study results towards a smaller overall impact and smaller overall improvement potentials relative to the result if these impacts had been quantified. It is not expected that inclusion of these impacts would change the overall conclusions of this study.

Conclusions

From the results, it is particularly interesting to note that:

- The consumption of meat and dairy products constitutes only 6% of the economic value of the total final consumption in EU-27, while contributing 24% of the environmental impacts (with a large variation between impact categories, e.g. from 6% for terrestrial ecotoxicity to more than 35% for eutrophication, nature occupation and aquatic ecotoxicity).
- The monetarised environmental impacts (externalities) are of considerable size compared to the private costs of the products (from 34% of the private costs for pork to 112% of the private costs for beef). The large uncertainty on the monetarisation implies that this proportion can be an order of magnitude smaller or larger.
- The aggregated (monetarised) result is dominated by three impact categories: Nature occupation (49%), Respiratory inorganics (23.5%) and Global warming (22.5%), thus leaving only 5% for all other impact categories. Using as an alternative the weights from Ecoindicator99 does not alter this picture, although slightly shifting the relative importance between the three large impact categories.
- The four main product groups (dairy, beef, pork and poultry products) contribute respectively 33-41%, 16-39%, 19-44%, and 5-10% to the impact of meat and dairy products consumption in EU-27 on the different environmental impact categories.
- Per kg slaughtered weight, there is a clear difference between the three types of meat, with beef having 4 to 8 times larger environmental impacts than poultry and up to 5 times larger than pork. These differences are less pronounced when comparing the environmental impact intensity (impact per Euro spent) of the three types of meat, where pork generally has the lowest impact intensity (down to 40% of the impact of poultry and 23% of the impact of beef), with the exception of aquatic ecotoxicity where pork production contribute with high copper emissions.

When all the identified environmental improvement potentials are taken together, the total improvement amounts to a reduction of 17% for nature occupation, around 25% for global warming and respiratory inorganics, 31% for acidification and terrestrial eutrophication, 43% for aquatic eutrophication, to 68% for aquatic ecotoxicity (when rebound effects and synergies have been accounted for). Since the first three impact categories make up 95% of the aggregated (monetarised) environmental impact, the aggregated improvement potential amounts only to 21% of the total environmental impact of meat and dairy products in EU-27 (and significantly less if rebound effects were not accounted for). Noting that the aggregated impact from meat and dairy products amount to 24% of the overall impact of EU-27 total final consumption, this implies that *after* all improvement options have been successfully implemented, the impact from meat and dairy products would still amount to 19% of the aggregated impact of EU-27 total final consumption. This seems to suggest that large reductions in the overall impacts from meat and dairy products cannot be obtained from the identified improvement options alone, but will require targeting the level and mode of consumption as such. One of the proposed improvement options may be applicable also for this purpose, namely household meal planning tools. While it may also be relevant to increase the production and/or consumer costs through environmental taxes, to internalise the identified externalities, the relatively low price elasticity of food products suggests that such a measure alone would not provide the desired proportional reduction in impacts.

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Life cycle greenhouse gas emissions from Brazilian beef

C. Cederberg[1], K. Neovius[2], D. Eivind-Meier[1], A. Flysjö[1], Ulf Sonesson[1]
[1] SIK The Swedish Institute of Food and Biotechnology, PO Box 5401, 402 29 Göteborg, Sweden, christel.cederberg@sik.se

[2]Karolinska Institute, Social Medicine, Norrbacka, 171 76 Stockholm, Sweden

Keywords: Beef production, Brazil, greenhouse gas emissions, land use changes, deforestation

Abstract

The growth of Brazilian beef production during the last decade, corresponding to two million tonnes carcass weight equivalents (CWE), has been export-driven; domestic consumption has not increased. Brazil´s growing importance for the global beef market is exceptional and its total export has increased by almost 600 percent during the last decade. This study suggests that approximately half of the production increase during the past ten years has taken place in the states outside the Legal Amazon and approximately half has occurred in the nine states of the Legal Amazon. The production increase in the Legal Amazon is partly an effect of a growing cattle herd and an increasing pasture area. CO_2 emission from land use changes (deforestation) is the predominant contributor of GHG emissions from beef production in Brazil and is explained by pasture expansion into forest in the Legal Amazon. Methane emissions are higher than estimated in studies of European beef production which is explained by higher slaughter age and longer calving intervals for the cows.

Introduction

During the past years there has been a rising awareness of the many environmental impacts caused by a rapidly growing global production and consumption of animal products. According to the FAO-report “Livestock´s Long Shadow” (Steinfeld *et al*, 2006), the global livestock sector is one of the top two or three most significant contributors to some of the most serious environmental problems of today, on every scale, from local to global. In the FAO-report it was estimated that the world´s production of meat, milk and eggs are the cause of ~18 percent of total global greenhouse gas (GHG) emissions.

There is a lack of environmental assessments of meat production in tropical countries (almost all studies are done for temperate conditions) and this knowledge gap is a motive for this study. Brazilian beef production is growing rapidly and in only one decade, Brazil has become the major beef exporter of the world. Over the last decade, there has been a strong expansion of the cattle herd in the Legal Amazon, from 39 to 56.5 million heads, and this region now holds a third of the Brazilian cattle population. The Legal Amazon is an administrative unit (5.5 million km^2) which include the nine Brazilian states: Acre, Amapá, Amazonas, Pará, Rondônia, Roraima, Tocantins, Maranhão and Mato Grosso. According to Margulis (2004) cattle ranching enterprises now occupy nearly 75 percent of the deforested areas in the Amazonia region. Fearnside (2008) conclude that cattle ranchers are key actors in Amazonian deforestation and responsible for most of the clearing.

Pasture is the overall dominant feed in the beef production and only five percent of the slaughtered animals in 2006 were raised in feed-lots. Of the total pasture area of 178 million hectares (Mha), 100 Mha is planted grass and 78 Mha is native vegetation (so called rangeland). Overgrazing and lack of nutrient replacement leads to pasture degradation and according to de Oliviera *et al* (2004), the land area occupied by degraded cultivated pastures in the tropical region of Brazil is estimated to be over 25 Mha. Pasture degradation leads to a substantial loss in productivity and in order to compensate for production loss, farmers usually incorporate new areas of native savannas and forests.

The overall aim of this study was to quantify the total greenhouse gas (GHG) emissions from Brazilian beef production. The Brazilian beef production was analysed with a top-down national perspective. Besides from the potential impact on global warming, also use of land and energy was analysed, but are not discussed in this paper, for a full report see Cederberg *et al* (2009). According to Steinfeld *et al* (2006), land use changes caused by an expanding livestock sector in South America are of great importance to GHG emissions and habitat destruction, and therefore a deeper analysis of deforestation related to Brazilian beef production was carried out.

Method

Scope of the study

The study dealt with all the phases as shown in Fig. 1 including production of materials and energy used. GHG emissions from land use transformation caused by the expansion of pasture into forestland were also included.

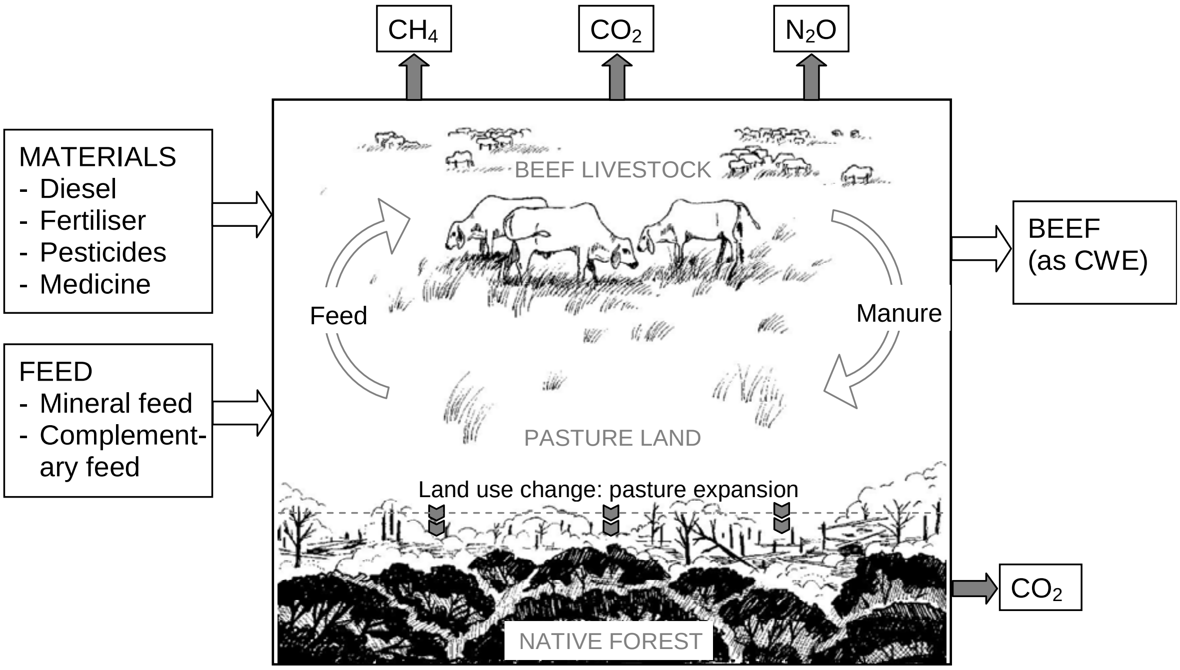

Fig. 1: The production system studied for the analysis of Brazilian beef production ending up as 1 kg of carcass weight equivalent (CWE) at the farm-gate

Data collection

Data on resource use and emissions from Brazilian beef production were collected from statistical sources, recent published scientific literature and through frequent contacts with Brazilian researchers in the fields of agriculture and environment. Assessments of emissions of methane and nitrous oxide rely on that livestock population, fodder intake and production systems are being well characterised and collected data were insufficient to fulfil all requirements needed for a complete analysis. In a country of Brazil´s size and with its large and expanding livestock production, there is an urgent need of more detailed basic data on beef production systems in different climatic regions. This was concluded already when the first inventory report on GHG emissions was compiled (early 1990s) (EMBRAPA 2002) and is even more inquired for today in the light of Brazil´s growing importance to the global beef production.

Land use changes

The GHG emissions from deforestation were calculated with a method referred to as “net committed emissions (NCE)”, which calculates emissions as a result of net difference in carbon stock of the original and replacing vegetation. This method differs from the more commonly used cross-sectional method (“annual balance”), by means of including all emissions caused by the deforestation act, no matter when in time they occur. When forestland is cleared for pasture or cropland, the fate of the land can go into different directions. Land use change patterns in the Brazilian Amazon are dynamic and complex and involve different cycles of clearing, cultivation, grazing and secondary forest re-growth (Foley *et al*, 2007). In order to cope with the frequent land-use transitions following deforestation, a Markov model based on different states (here represented by land-use) was chosen when calculating NCE. A Markov model of agricultural land use in Brazilian Amazonia was developed by Fearnside (1997) with transition probabilities between different land uses.

The initial aboveground biomass was adjusted down by 7 percent carbon removed in the biomass before clearing the forest for agricultural purposes. In analyses with a product perspective (for example LCAs or carbon footprinting of products) the GHG emissions from one single year of deforestation must be distributed over time for the production generated from the land use (pasture or cropland) following the clearing. We choose to distribute the calculated GHG emissions over 20 years and this time-period is in agreement with the proposed EU-directive on the promotion of the use of energy from renewable sources, stating that annualised emissions from carbon stock changes caused by land use change, shall be calculated by dividing total emissions equally over 20 years (EC 2008).

It was estimated that milk production used approximately 20 Mha pasture in 1996 which was slightly more than 10 % of total pasture area in Brazil. Dairy production is concentrated in the south and south-east regions (close to consumer markets) of Brazil and of minor importance in the Legal Amazon. Fearnside (2008) states that only in limited areas of Brazilian Amazonia, milk production and processing are activities that drive deforestation. However, we did an allocation of use of pasture between beef and milk production of 90 % to beef and 10 % to milk.

Results

Export-driven beef growth

A decade ago, Australia, the USA and the EU were the major beef exporters but a significant shift has taken place on the global beef market and in 2004, Brazil became the largest exporter of beef. Brazil´s growing importance for the global beef market in recent year is exceptional and its total export increased by almost 600 percent during the last decade. We concluded that the overall growth of Brazilian beef production during the last decade has been export-driven, since the domestic consumption has not changed or even slightly been reduced (ANUALPEC/FNP 2006) (Fig. 2).

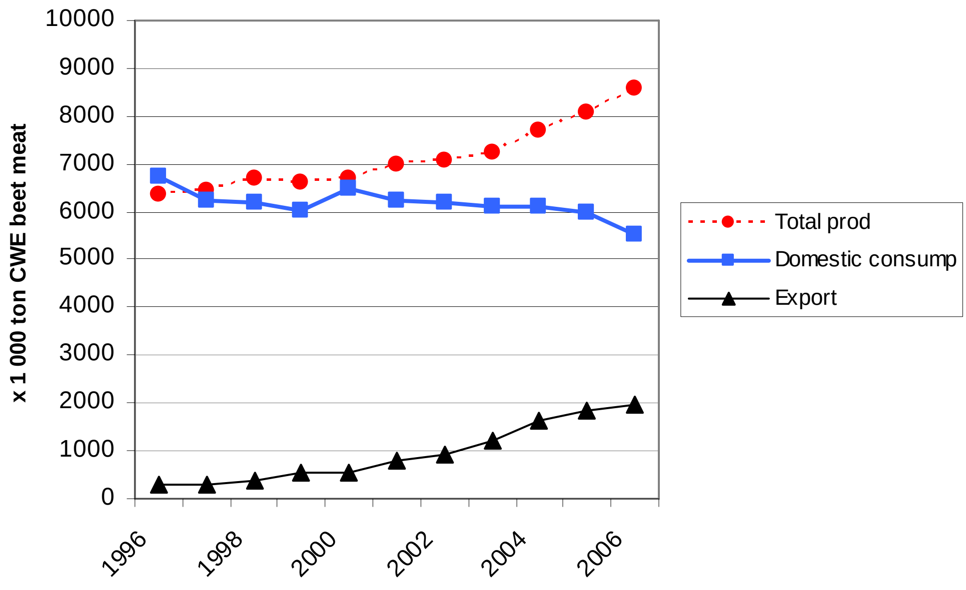

Fig. 2: Total beef production, internal consumption and beef export (x 1000 ton carcass weight equivalent, CWE beef meat) in Brazil 1996 - 2006

Growth of beef production and land use

In Tab. 1, changes in beef production over the last decade are analysed using data from the period 1997 – 2006 (ANUALPEC/FNP 2006). Approximately half of the production increase in Brazil during the past ten years has taken place in the states outside the Legal Amazon. This was done without increasing the number of cattle. Increased animal productivity with lowered slaughter age is the most plausible explanation for this positive development. Approximately half of the production increase, has occurred in the nine states of the Legal Amazon and here the increase seem to be an effect of improved animal productivity as well as an increase of the total cattle population.

Tab. 1: Increase of beef production (10^6 tonnes CWE) in total Brazil, the nine states of the Legal Amazon and Brazil except Legal Amazon

	Brazil total	Legal Amazon	Brazil except the nine states of Legal Amazon
		10^6 tonnes CWE	
1997	6,444	1,096	5,348
2006	8,582	2,021	6,561
Increase, 1997-2006	+2,138	+925	+1,213
Share of increase, 1997-2006		**0.43**	**0.57**

Source: ANUALPEC/FNP 2006

Also, the overall change in agricultural area between 1995 and 2006, shows a remarkable increase of almost 30 Mha. Approximately 75 percent of this growth has taken place in the nine states of the Legal Amazon (IBGE 2007).

CO_2 from land use changes

During the 20-year period from 1987 – 2006, ~36.7 Mha of native forest have been cleared in the nine states of the Legal Amazon, corresponding to on average ~1.8 Mha per year (INPE 2008).

We estimated the expanding beef production in the Legal Amazon to be the source of approximately 700-800 $\cdot 10^6$ tonnes CO_2 emitted annually from deforestation for the time-period 2005 – 2007. This can be compared with the estimate in the FAO-report ”Livestock´s Long Shadow” concluding that livestock-related land-use changes globally may emit 2 400 $\cdot 10^6$ tonnes CO_2 per year (Steinfeld *et al*, 2006). The estimation of FAO was based on the ongoing pasture expansion into forest by an annual average of 2.4 Mha and the cropland expansion into forest by an annual average of 0.5 Mha in South America. The results presented herein imply that about one third of the emissions caused by deforestation for gaining more land for pasture and feed crops in South America according to the FAO-study could be caused by expanding beef production in the Brazilian Amazon.

Methane emissions

The methane emissions caused by enteric fermentation for the year 2005 were calculated with ANUALPEC/FNP (2006) statistics of cattle population (~149 million heads, dairy cows not included). Using the most recent emission factors (EFs) suggested by Lima *et al* (2007) gives an average emission of 0.80 kg CH_4 kg CWE^{-1} and the EFs according to IPCC (2006), Tier 1 gives an average emission of 0.85 kg CH_4 kg CWE^{-1}. This is approximately 40 percent higher that estimated methane emissions from meat production in suckler-cow beef systems in Sweden. The most important explanation for the higher methane emissions per kg of product is an overall lower productivity in Brazilian beef production systems. The slaughter age is on average around three years compared to 18 – 24 months in Sweden. Late weaning, often an effect of poor pasture and nutrition, leads to longer inter-calving intervals and thereby reducing overall calf production of breeding cows in the herd. Calving intervals are around 20 months, compared to approximately 12 months in Sweden/Western Europe.

Discussion

CO_2 emission from land use changes (deforestation) is the predominant contributor of GHG emissions from beef production in Brazil and is explained by pasture expansion into forest in the Legal Amazon.

There is an urgent need to reduce deforestation rates and the increasing trend from 2000 to 2004 is now broken. In 2007, an area of ~1.15 Mha was estimated to be deforested by the INPE (2008) and so far in 2008 (up until August), the deforestation rate is 567 600 ha. During the spring of 2008, the Brazilian government launched the operation *Arco de Fogo* (Arc of Fire) to stop deforestation in the Legal Amazon. Military forces as well as government agencies are involved and focus on illegal extraction and sale of timber in the region.

Most of the beef export origins from states in the south- and central-east of Brazil, from regions where there is no large-scale deforestation. These exporting states have an advantage on the market compared to states in the Amazon region because of better infrastructure, more modern slaughter-houses and they have had a longer time of Foot and Mouth Disease-free status and thus have been allowed to export beef to the EU and other important markets. Although the production from the beef-exporting states not directly is the source of CO_2 emission from deforestation, it is our conclusion that all the Brazilian beef production must carry the burden of emissions caused by land use changes. The Brazilian domestic consumption of beef has not increased during the last decade and the strong growth is driven by an increased demand on the export market. Beef produced in the Legal Amazon is to a great extent exported to the south and south-east regions of Brazil (Arima *et al*, 2006) where the most important domestic consumer markets are. In practice this means that the increasing beef production in the Legal Amazon during the last decade has been used to compensate for the beef produced in the south- and central-east that is no longer available for domestic consumption since the export market has expanded so much.

Conclusion

Poor pasture management is an important environmental hot-spot in Brazilian beef production. Recent estimations say that more than 25 Mha of planted pastures can be degraded in various stages, thus leading to low productivity followed by need for new pasture land. The ongoing land expansion could be substantially reduced if pasture land were better maintained. Pasture degradation can be prevented by maintenance fertilisation and avoidance of high stocking rates, especially in dry periods. Methane emissions can be reduced by improving livestock performance, e.g. by shortening calving intervals and lowering slaughter age but also by improved pasture management. Overall productivity would benefit from using more supplementary feed as complement to pasture in dry periods and also by increasing the use of more intense production forms, such as feed-lots and integrated livestock-cropping systems.

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LCM in agriculture: enhancing the self-responsibility of farmers

M. Alig[1], G. Gaillard[1], G. Müller[2]
[1]Agroscope Reckenholz-Tänikon Research Station ART, CH-8046 Zürich, Switzerland, martina.alig@art.admin.ch.
[2]ArGe Natur und Landschaft, CH-6133 Hergiswil

Keywords: LCM in agriculture, communication of LCA-results, LCA on farm-level

Abstract

Agriculture has manifold impacts on the environment and plays an important role in the environmental impacts of food chains. Although in Switzerland all the farms obey the same laws, there is still a high variability in the impacts of farms on the environment. This indicates that further optimisations are possible. Therefore, the farmer is an important factor. In order to improve the environmental impacts of his farm, he needs information where those impacts occur and why. Life cycle assessment is an instrument which can provide the farmer with such information. In a pilot project with 200 farmers we are developing a concept on how to communicate life cycle assessment results to farmers, so that they are able to understand their results and draw conclusions for the management of their farm.

Our results allow us to identify which means of production contributed most to an environmental impact and give the farmers important hints about their environmental profile. It is shown that for each environmental impact different factors are important. The important factors also vary between the single farms. To further improve the environmental impacts of farms a detailed, holistic and individual analysis is necessary. The confrontation of the LCA results with economical figures as well as with production parameters promotes a holistic, integrating view on environment, productivity and economy. Most farmers proved willingness to deepen the analyses of the feedback and agreed with the necessity of LCM on farms.

Introduction

Farming shapes many regions in Europe and often plays a great role in the environmental impact of food chains. In most European countries governments try to limit the harmful impacts of agriculture on the environment by enacting regulations and bans. Indeed, such initiatives enable to implement minimal environmental standards. On the other side, such laws have the disadvantage to deprive the farmers of their self-responsibility - they just carry out and do not decide themselves.

A LCA-study of 50 Swiss farms has shown that - although all farms obey the same laws - there is still a high variability in the environmental impacts of farms (Gaillard & Rossier, 2001). This indicates that further optimisations are possible. Therefore, the farmer is a very important factor. With his management decisions he decisively influences the environmental impacts of his farm (Nemecek *et al*. 2005). But until now, a farmer has no feedback on his individual impact on the environment, what would be very important if he wanted to improve. In order to be able to minimize the negative impacts of his farm on the environment, he has to know where the environmental impacts of his farm occur and what the reasons for those impacts are. At the same time, it is a big advantage for a farmer to know where he performs well and to be able to prove this with data in order to satisfy the increasing interest in environmentally friendly production of retailers and consumers.

An instrument which provides the farmer with such information is Life Cycle Assessment (LCA). Unfortunately, this instrument is rather complex and not easy to understand for a non LCA expert. But for an efficient and sustainable life cycle management it is essential that the farmers understand the method of life cycle assessment and are able to interpret their results. So how can these inherently complex topic be communicated to non LCA experts? In a pilot project with 200 farmers we are analysing where the major impacts on the environment of a farm in real situations occur and how these results can be communicated to the farmers in such a way that they are able to understand the results and transfere them into practice.

The project “Life cycle assessment – farm accountancy data network”

Based on specific criteria considering farm type, region and production system (integrated / organic) we recruited 200 farms and supplied them with software for recording the production data. The software is based on existing, commercial farm management software (AGRO-TECH, © Agridea) and can also be used for various other records a farmer must keep according to the legislation. During the period of the project, the use of the software is free of charge.

The participating farmers assess their production data during three years (2006 – 2008). Each year, a detailed life cycle assessment is calculated for every farm and each farmer receives a report with his results. On developing the reports with the results we could draw on experiences gathered in a previous project in which we had elaborated reports with LCA-results for three farmers (Baumgartner et al, 2006). Based on the positive feedbacks of those farmers we developed the effective reports and tested it in workshops with some of the participating farmers. Their feedback was subsequently integrated in the definitive reports for the first year, which will be presented in the following.

For a maximum explanatory power we chose to analyse five environmental impacts relevant for agriculture: energy demand, global warming potential, eutrophication and aquatic and terrestrial ecotoxicity. The LCA results are expressed at farm level and for the main production branches. Thereby we compare each real farm with a so-called reference farm (i.e. a theoretical farm representing the mean environmental impact for the type of farm analysed (concept in Nemecek et al, 2004)). The comparison with the reference farm was chosen to allow the comparison with an average Swiss production. Like that, a farmer knows where the average is and can also be better than the reference farm, which gives him a lot of positive motivation. With this system, one can also imagine a flexible benchmark, with a steadily improving reference farm as the real farms as a whole and with them the Swiss average improves.

The participating farms are at the same time reference farms of the farm accountancy data network, so the ecological data can be put in perspective with the accounting data and combined ecological-economical analyses are possible.

In order to improve the understanding of the results we held large-scale workshops with the farmers after they had received their results. In those workshops the farmers again had the possibility to give us their feedback about the result-reports. These feedbacks will be integrated in the reports of the next year.

Results of the first year

The first results confirmed the findings of the pre-study; there is a very high variability between the different farms (Fig. 1).

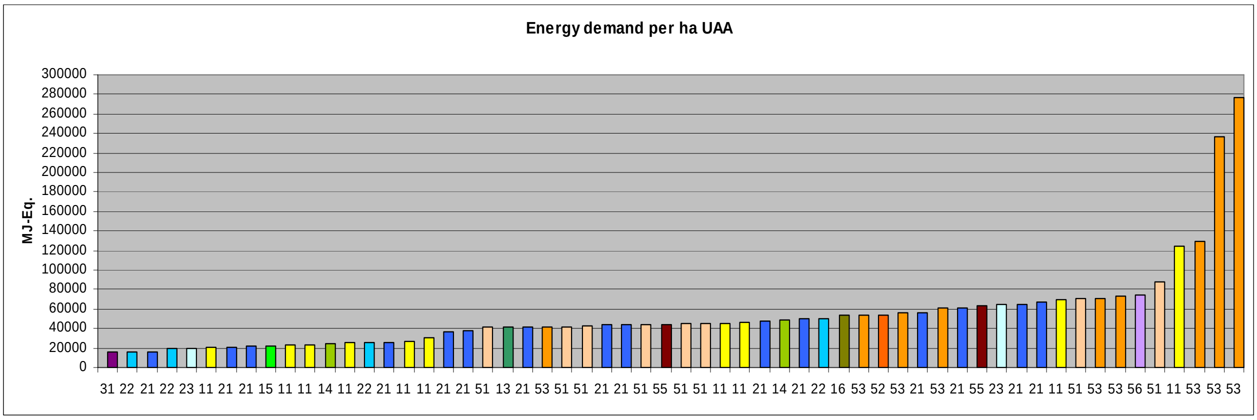

Fig. 1: Energy demand per ha UAA of 62 Swiss farms. Each number and colour corresponds to a certain type of farm (see Tab. 1).

Tab. 1: Farm types of the participating farms

Farmtype	Description	Farmtype	Description
11	arable farming	23	other cattle
13	vegetable cultivation	31	horses/goats/sheep
14	fruit cultivation	51	dairy farm / arable farming combined
15	viticulture	52	suckler cows / arable farming combined
16	other cultures	53	pigs and poultry / arable farming combined
21	dairy farm	55	dairy farms / other combined
22	suckler cows	56	cattle / other combined

The farm with the lowest energy demand used only 5 % of the energy the farm with the highest energy demand used. The three farms with the highest energy demand are all combined farms (cropping and pigs or poultry). Since the results are expressed per ha UAA, the total surface of the farm could influence the results: a specialised farm with a practically soil-independent production must look very bad in a comparison based on results per ha UAA. But in our sample, there are no farms with a soil-independent production, also the three farms with the highest energy demand per ha UAA do some arable farming and have no smaller surface than the rest of the farms considered. The type of farm as well does not explain the full difference; there are other farms with pigs or poultry which use much less energy. And also within a certain farm type there is a high variability in the energy demand: the dairy farm with the highest energy demand used over three times more energy than the dairy farm with the lowest energy demand (Fig. 2).

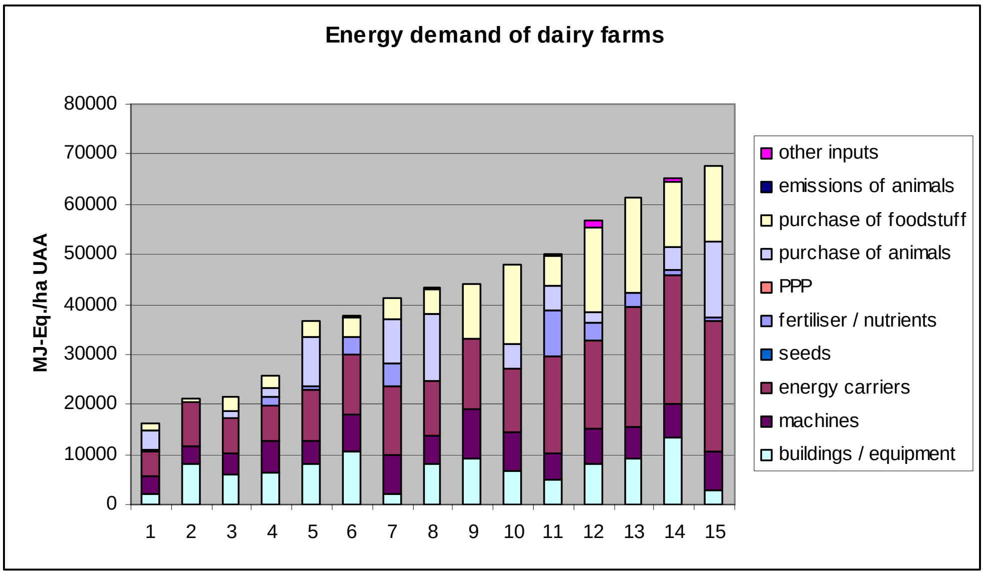

Fig. 2: Energy demand per ha UAA of Swiss dairy farms

The major explanatory factors for the energy demand are the utilised energy carriers, the energy use for the production (resp. construction), maintenance and disposal of the existent machines and buildings and the energy input for the production of the purchased foodstuff. Depending on the farm, also other burdens from the upstream chains (energy consumption for the production of the used mineral fertilisers or during the upbringing of purchased animals) can play a role. The share of the different input groups in the total energy demand as well as the absolute energy demand per input group varies depending on the farm. The reasons for these differences are manifold: One farm lies in the mountains and therefore needs a lot of diesel for driving its tractor up and down, the other farm has very old and inefficient tractors, and the third one needs a lot of current for some special devices in its stables. On that account, every farm needs to be analysed individually to exactly determine the reasons for a high result in an environmental impact.

Regarding eutrophication, the picture is a different one (Fig. 3). Now the major explanatory factors are the use of fertilisers and the emissions from animals (ammonia and nitrate in stable and pasture). For some farms, also the burdens from the upstream chains (emissions during the upbringing of purchased animals or the cultivation of the purchased foodstuff) contribute significantly to the eutrophication.

The variability between the farms is still high, but the ranking of the farms is not the same as with the energy demand. Some farms which had a low to medium energy demand have a very high eutrophication, and farms with a high energy demand have a low eutrophication. What is striking to note is that the farm with the lowest energy demand has also the lowest eutrophication, and the farm with the highest energy demand has also the highest eutrophication.

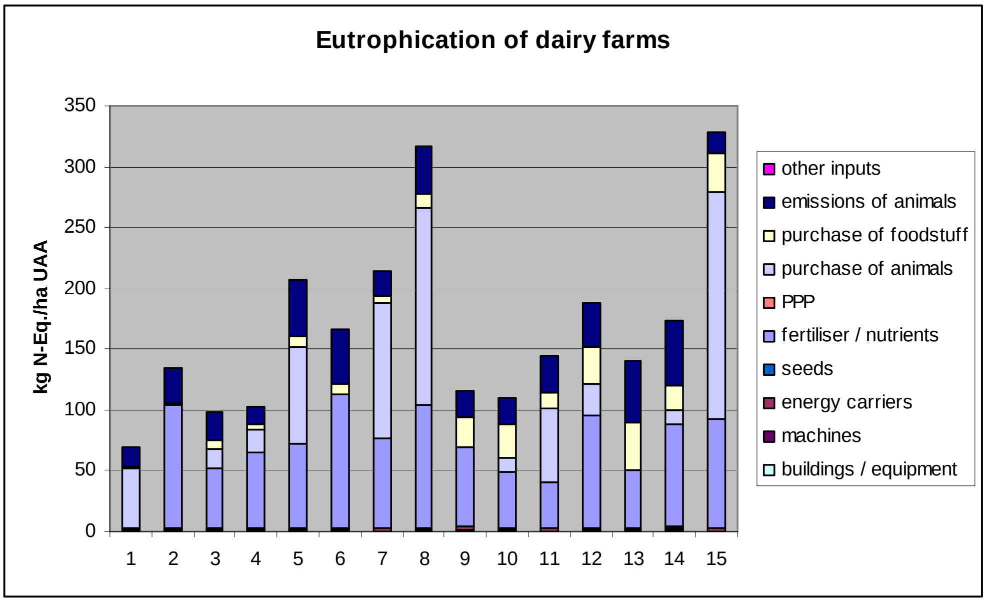

Fig. 3: Eutrophication per ha UAA of Swiss dairy farms

Feedback to the farmers

The report to the farmers is divided into 3 chapters. First we give an introduction, where the most important aspects of life cycle assessment are shortly described and important terms are explained. Then the main descriptive data of the farm and the reference farm used as benchmark is given. This allows the farmer to see with what he is being compared and gives him important hints for the interpretation, as he can recognise where there are differences in the production.

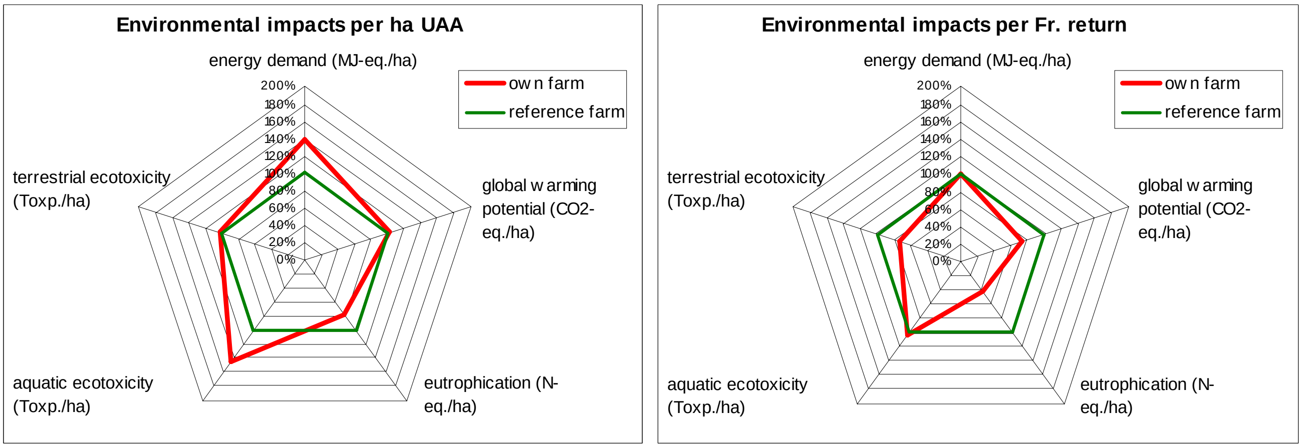

Fig. 4: Overview of the environmental impacts of a farm per ha UAA and per Swiss-Fr. return as presented in the reports for the farmers

In the third chapter the results of the LCA are presented. First we give an overview of the results of the whole farm (Fig. 4). All results are presented relative to the three functional units “usable agricultural area”, “digestible energy” and “gross profit”. The functional unit “usable agricultural area” represents the intensity of the production, whereas the functional unit “gross profit” is targeted on the economic performance. The functional unit “digestible energy” is related to the energy a human being can digest and represents the total production of the farm with all its different products converted to their energy content available for man. The analyses for a single kind of product, e.g. cereals, beef or milk, is made later by the analysis per product group, where the environmental impacts are expressed per kg product (see Fig. 6).

After the overview of the farm, the share of the different means of production in the environmental impacts is shown for all five environmental impacts analysed (Fig. 5). Finally, the environmental impacts are analysed by the three most important products of the farm (Fig. 6) and expressed by kg product.

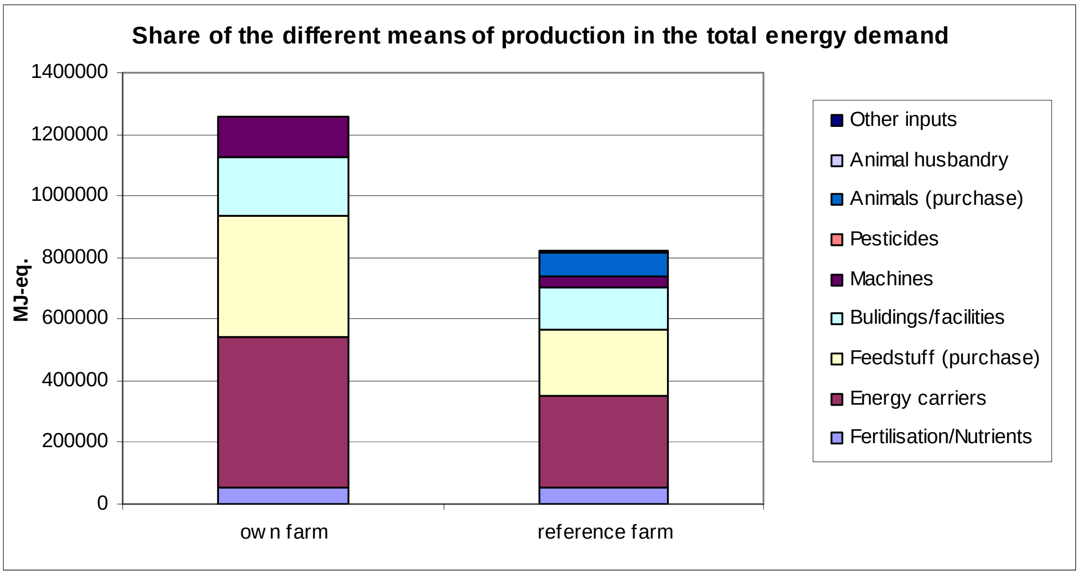

Fig. 5: Share of the different means of production to the energy demand, one of the 5 important categories detailed in the reports for the farmers

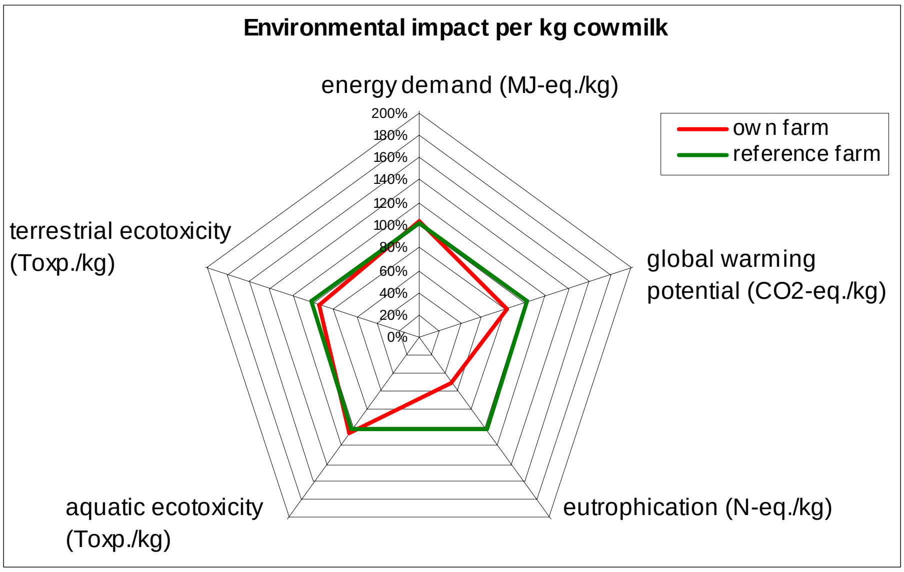

Fig. 6: Environmental impacts for the most important product group of the farm, as shown in the report for the farmers

The combining of all this information, together with general data on farm production, shall allow the farmer to trace potential big impacts back to their cause. To help with the interpretation we provide the farmers with examples and give them lists of possible reasons for bad results in a certain environmental impact. At the end the results of the farm in the categories “energy demand” and “eutrophication” are compared with other farms of the same farm type. This shows the farmer the position of his farm within the other participants and gives him an idea about the possible ranges of the results. Like this, he shall be inspired to take action in order to belong to the best farms of his stratum.

Conclusion

Considering the high variability of the environmental impacts of the farms it is clear that further improvements are possible. But the question how these improvements can be achieved is not an easy one. For each environmental impact different factors are important. The important factors also vary between the single farms. To further improve the environmental impacts of farms a detailed, holistic and individual analysis is necessary. Thereby the results have to be communicated to the farmer and he has to understand them, so that they can be transferred into practise. The concept of the reference farm helps by giving an orientation to the farmers and makes it easier for them to range their results. But the heterogeneity observed within one type of farms makes the comparison between the real farm and the reference farm sometimes difficult. Therefore the comparison of the real farms was also very important and gave the farmers a lot of motivation.

Our results allow us to identify which means of production contributed most to an environmental impact and give the farmers important hints about their environmental profile. In addition, the LCA results are confronted with economical figures as well as with production parameters like the output of digestible energy. This allows us to set the environmental impacts in perspective with aspects of income generation and physical productivity and promotes a holistic, integrating view on environment, productivity and economy.

From the farmer’s point of view, also the results per product group were very interesting, as this functional unit corresponds best to the commercial view most farmers have regarding their production. But this approach requires many allocations, which is not always easy to understand and leads to a certain simplification. Additionally, it only makes sense if the farms compared have the same products and a very similar structure. Therefore the results per product group can only be seen as an additional way of illustrating the results and have to be analysed in the context of the results for the whole farm.

The workshops showed that the LCA feedbacks provoke serious reactions, especially in case of rather negative results. Some figures are hardly to be traced back to clearly defined causes and the possibilities for the farmers to react in a direct and efficient way are restricted. On the other hand, most of the farmers proved willingness to deepen the analyses of the feedback and agreed with the necessity of LCM on farms. In the following two years we will further improve the feedback, i.e. in the domain of benchmarking.

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A simplified LCA tool for Environmental Product Declarations in the agricultural sector

P. L. Porta, P. Buttol, L. Naldesi, P. Masoni, A. Zamagni
ENEA-Italian National Agency for New Technologies, Energy and the Environment
Via Martiri di Monte Sole 4, 40129 Bologna (Italy).
pierluigi.porta@bologna.enea.it

Keywords: Simplified LCA; SMEs; Environmental Product Declaration; EPD.

Abstract

Different systems and certification programmes of ISO type III labels have been developed all over the world. However, the diffusion of EPDs at SMEs is quite difficult due to complexity and costs of the EPD preparation and certification process. In the framework of the LIFE-Environment project "Ecoflower Terlizzi - Demonstration project for the Environmental Product Declaration: the flowers of Terlizzi and the local eco-label", this problem was addressed on the basis of the authors' experience in the development of simplified tools for SMEs. Starting from a first version of an on-line tool for screening LCA, eVerdEE, a second version was developed suitable for applications in the agricultural sector and the related database was extended including datasets referring to flowers production. A simplified methodology has been developed, which includes the definition of an EPD programme and the implementation of a procedure to automatically generate the EPD starting from an eVerdEE study. To validate the methodological simplifications introduced, the eVerdEE results were compared with those of detailed LCA studies. The good results obtained are only a first step towards the validation of the entire approach. In this paper the methodological approach adopted is presented and discussed and suggestions for further activities are given.

Introduction

Environmental life-cycle-based labels are effective tools to provide consumers with information on the environmental performance of products and services and to increase their environmental awareness. In particular ISO type III Environmental Product Declaration (EPD) allows communicating the results of an LCA study and adding other significant environmental information that are not captured by the LCA study. It can also be used for communications within the supply chain and in green public procurement to provide customers with quantitative environmental information about the product. If we analyse certification systems and programmes that have been developed all over the world, we can observe that in general they result in EPD preparation and certification processes complex, quite expensive and time consuming. This is a barrier to the diffusion of EPDs at SMEs. Moreover, SMEs often experience difficulties to perform detailed Life Cycle Assessment (LCA) studies due to internal lack of competence and resources. Both these aspects can be a problem for the agricultural sector especially in Italy, where small family-enterprises prevail. Simplified tools and methodologies are then necessary to extend the use of LCA and EPDs. In the framework of the LIFE-Environment project "Ecoflower Terlizzi - Demonstration project for the Environmental Product Declaration: the flowers of Terlizzi and the local eco-label", funded by the European Commission (LIFE ENV/IT/000480), this problem was faced by making available to SMEs a simplified LCA tool, tailored to the specificities of the agricultural sector and linked with an automatic procedure for EPD preparation, and a simplified EPD programme, whose compliance with the ISO 14025 standard was certified by a third party.

Method

Simplified LCA

A simplified LCA, which can reduce methodological complexities and resources investment, represents a way to extend LCA usability among SMEs but it has to keep sound scientific basis in order to guarantee reliability and robustness of results. For this reason starting point of ENEA's research is the consideration that simplified LCA can assure quality and credibility only when results from detailed LCAs are available, in order to guarantee comprehensive data of high quality and detail, understanding of uncertainties and identification of the critical aspects of the life cycle. The method is based on a sectoral approach, organised as follow:

- Sector study, based on detailed LCAs of specific product chains, with the involvement of all stakeholders and sectoral competencies;
- Sector specific database, with data of the commonest processes, materials and components of the sector analyzed, fully integrated within the simplified LCA tool;
- Simplified LCA procedure, validated against the results of detailed LCAs.

The implementation of these three main steps is described in Fig. 1, grey background.

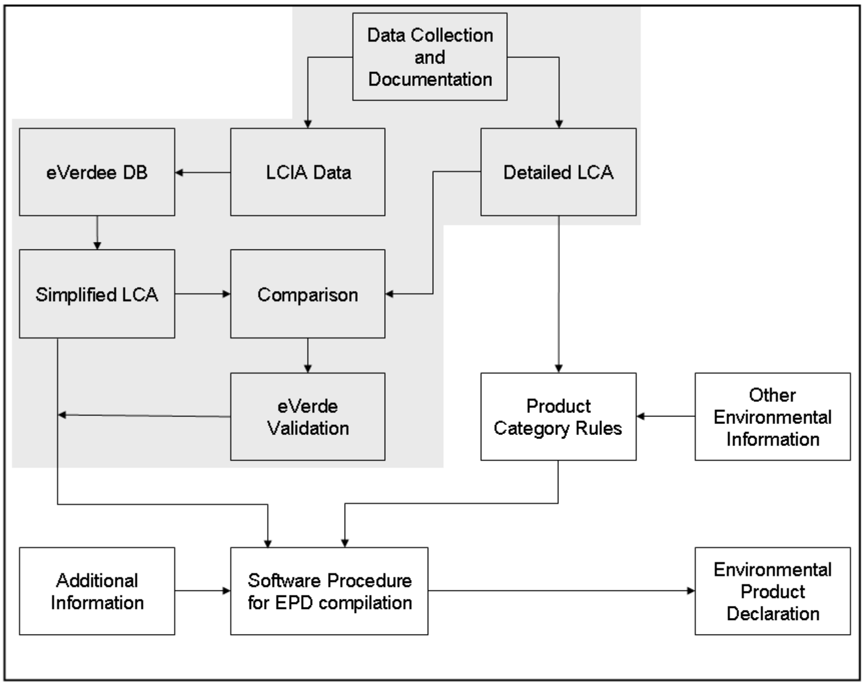

Fig. 1: Simplified scheme of the EPD process using eVerdEE.

Firstly, data are collected at enterprises and their suppliers in order to develop the LCA model. In Ecoflower project data collection involved enterprises which produce rose's stems and cyclamens with different production techniques. Data concerning the production of materials used during cultivation (fertilisers, pesticides, chemicals, etc.) and the construction of infrastructure and machinery (materials for greenhouses, pumps, tubes, etc.) were found in the open literature or in commercial DBs.

Data are used to carry out a detailed LCA study and to populate the eVerdEE database by using the proprietary software DIM (Data Input Manager), which stores data of processes and materials and calculates impact assessment data. eVerdEE database includes general and sector specific LCIA (Life Cycle Impact Assessment) data, i.e. impact assessment data of processes obtained from product chain studies. The quality of the DB is guaranteed by the methodological approach of the product chain studies which involve stakeholders, sector-specific technical competencies and peer review by external experts. A procedure of data documentation has been developed, based on ISO/TS 14048: 2002, in order to guarantee transparency and good quality of the whole database.

The database is the core of eVerdEE tool, on-line software for simplified LCA (Zamagni *et al*., 2005). The tool offers predefined options that help users to define goal of the study and system boundaries and examples to support the definition of functional unit, reference flow and allocation procedures. For each step of the inventory predefined forms are offered (Fig. 2), where users input their quantitative data referred to the reference flow, choose a corresponding entry from the database and evaluate the quality of the data they have input. Regarding the impact assessment, in eVerdEE elementary flows, impact categories, characterisation and normalization methods are predefined according to the screening characteristics of the methodology and selected on sound scientific bases. All those impact categories on which agreement exists in the scientific community have been considered, in particular: consumption of mineral resources, consumption of biomass, consumption of fresh water, consumption of non-renewable energy, consumption of renewable energy, climate change, acidification, eutrophication, photochemical oxidation. The characterization factors, when applicable, are extracted by the CML 2001 method. The Ozone Layer Depletion category has been replaced by the indicator "kg CFC-11", which is the only elementary flow of the eVerdEE list contributing to this impact category. Two environmental indicators (total waste and hazardous waste) have been added to take into account the production of waste during the life cycle of the product. Toxicity category has been excluded for two main reasons: i) consensus has still to be reached in the scientific community on how to properly deal with it in a context of LCA study; ii) the reduced number of elementary flows selected in eVerdEE cannot be fully representative of the toxicity category. The procedure for the calculation of the impact assessment uses the quantities input by users and the characterisation values of the eVerdEE database.

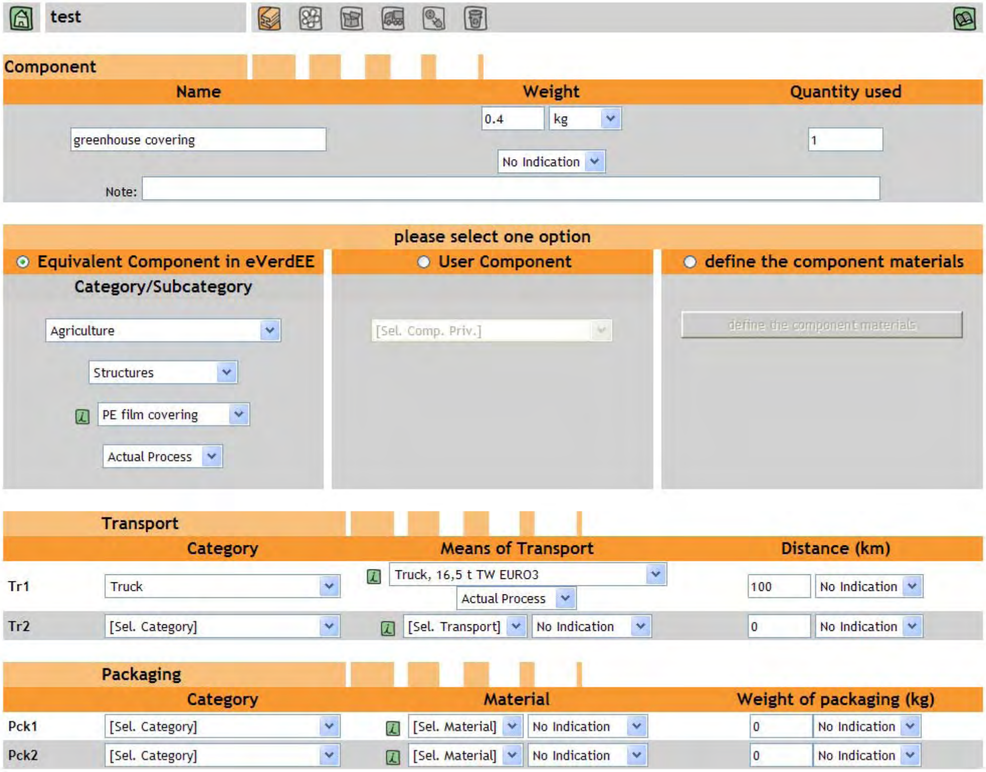

Fig. 2: Example of compiled predefined form of the eVerdEE inventory procedure

The application of the simplified methodology to agricultural products has required a revision of the life cycle model developed in the first version of the eVerdEE tool, which was designed specifically for the manufacturing industry.

The new model includes:

- A first phase (infrastructure and machinery), in which users define materials and components for construction and maintenance of the infrastructure (greenhouses, installations) and machinery (Audsley *et al.*, 1997);
- A second phase (Cultivation), in which users define every input and output of the cultivation process. To take into account those sector-specific emissions that are not included in the eVerdEE list of elementary flows (about 60 elementary flows of resources and emissions), it is possible to define 'user's emissions' in the inventory procedure. As 'user's emissions' are not classified and characterized in eVerdEE, they can be shown in the results only as an additional information;
- Packaging and distribution: packaging and transport related to the distribution of the product to the consumer;
- Use and end of life: materials and components used for the maintenance and repair of the product, energy consumption, outputs due to the use of the product, waste produced at the end of product's life.

An important step of the procedure is to validate the methodological simplifications introduced (model of life cycle, system boundaries, selection of elementary flows to be recorded, characterization methods used). A contribution to the validation of the approach adopted has been obtained by comparing the impact assessment results of detailed LCA studies on roses' stems (Russo *et al*., 2007) and of simplified eVerdEE studies. Comparison of applications in other production sectors is in progress.

Simplified EPD Programme

In the framework of the ECOFLOWER project, a simplified EPD programme has also been developed, which allows the use of eVerdEE for certification purpose. For this purpose a utility has been added to eVerdEE in order to produce automatically the EPD, in agreement with the programme proposed and with the two PCRs (Product Category Rules) developed for fresh cut flowers and fresh flowers in vase (Fig. 1). The automatic procedure extracts from the eVerdEE studies general information and LCA-based data, mainly functional unit, system boundaries, list of materials and substances, energy consumption, impact assessment results, waste production. As the use of pesticides is a relevant environmental aspect of the agricultural sector and toxicity is not assessed by eVerdEE, the problem is addressed during the EPD preparation when users are required to input into the EPD form quantitative data on all pesticides used during the life cycle of the flower together with the indication of the risk class (WHO, 2006) of each substance. Users are also asked for inputting into predefined forms the additional environmental information prescribed by the PCRs. Finally the declaration, which includes all data above mentioned, LCA-based and directly input by users, is printed for the third party verification. Both the LCA study and the EPD can be modified following the reviewer's comments. When the EPD is certified and the programme operator assigns the number of register, the final version of the declaration becomes publicly available and both LCA study and declaration are fixed and unchangeable.

Results

Data from four small enterprises which produce rose's stems have been used for the validation procedure. A detailed LCA model of the life cycle of rose's stems has been built by using GaBi software. Impact assessment results have been calculated using the CML2001 method. A simplified life cycle model has been built in eVerdEE; moreover a cut-off of 1% in mass for infrastructure and machinery materials has been assumed.

First of all, to verify that the cut-off rule chosen does not affect the results of the LCA studies, in the detailed LCA model we applied the same cut-off rule adopted in eVerdEE studies and we compared the results calculated in GaBi with and without cut-off assumption. As differences lower than 1% could be observed in the impact assessment results, we could go on with the comparison between the detailed LCA (GaBi) and the simplified LCA (eVerdEE) in order to validate the methodological simplification introduced in eVerdEE.

The only impact categories that can be directly compared are the following: acidification, eutrophication, climate change, photochemical oxidation and energy, as the sum of the two categories renewable and non-renewable energy (Fig. 3-7). The results of the impact categories consumption of mineral resources (calculated in eVerdEE) and abiotic depletion (calculated in GaBi) cannot be compared because the latter category classifies not only mineral resources but also fuels. Consumption of biomass and fresh water cannot be directly compared but careful aggregation of elementary flows of the GaBi inventory would be necessary.

As we can observe in s 3-7, the comparison of the results is good for all impact categories, with the exception of photochemical oxidation. Diesel oil for greenhouse heating gives the main contribution to this impact category with the emission of NMVOC into air. A detailed analysis of this data has shown that the amount of NMVOC emitted into air is different in the two studies because the process 'diesel oil production' comes from different sources in eVerdEE and GaBi database and not because of the simplification introduced in eVerdEE.

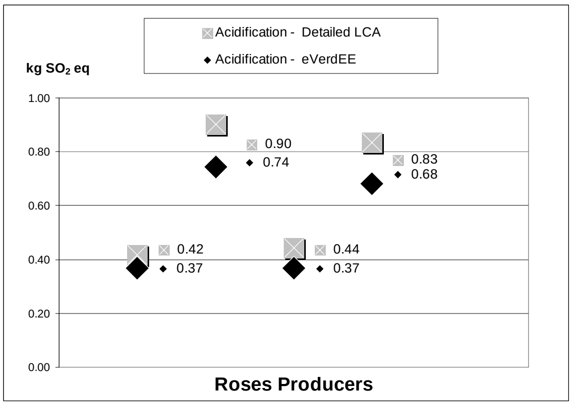

Fig. 3: Acidification results of detailed and simplified LCA studies

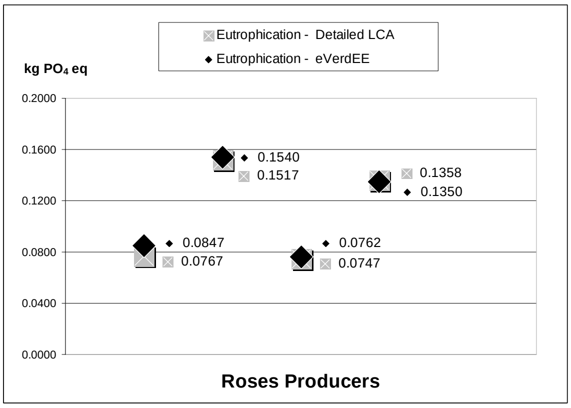

Fig. 4: Eutrophication results of detailed and simplified LCA studies

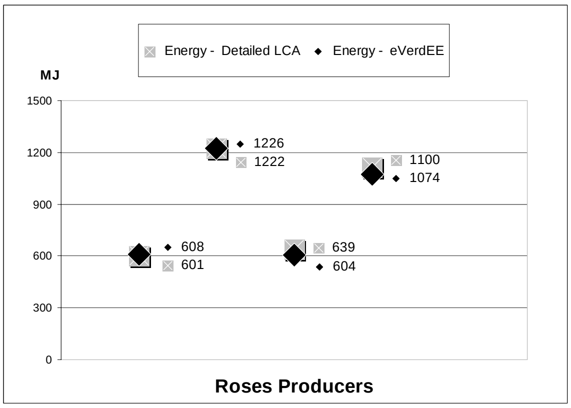

Fig. 5: Primary energy of detailed and simplified LCA studies

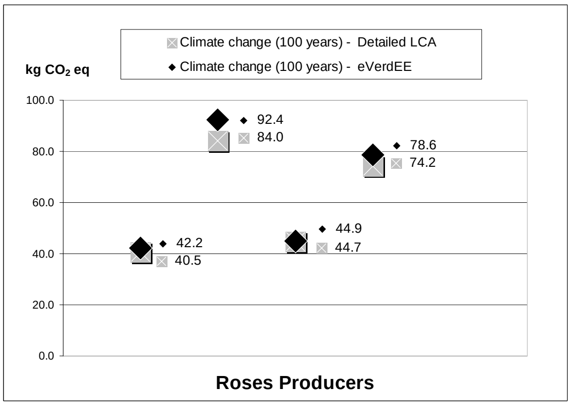

Fig. 6: Climate change results of detailed and simplified LCA studies

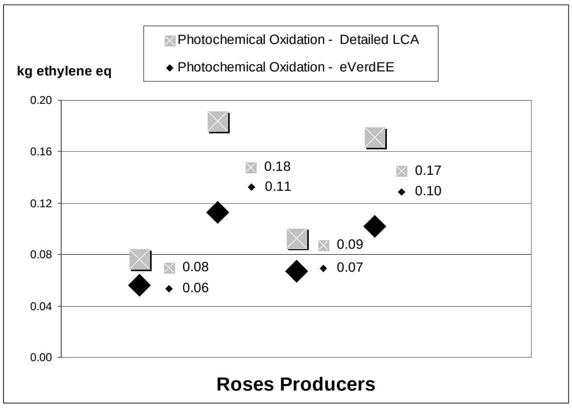

Fig. 7: Photochemical oxidation results of detailed and simplified LCA studies

Conclusions

eVerdEE is a tool for simplified LCA specifically designed to facilitate the adoption of LCA by SMEs. However, in the ECOFLOWER project, the possibility of using it for certification purpose has been investigated through the definition of a simplified EPD programme and the development of an additional procedure to produce an EPD on the basis of eVerdEE LCA studies.

In the case study here presented, the LCIA results from simplified LCA, obtained following the sectoral approach proposed, are in line with those from a detailed study, at least for the following impact categories: acidification, eutrophication, climate change, energy and, in a lesser extent, photochemical oxidation. These results are the first step towards the validation of the approach. The process has to be completed in two main directions: i) detailed analysis of the elementary flows of the categories which show the greatest divergences, in order to understand if the problem resides on the choice of the elementary flows; ii) analysis enlarged to other sectors, in order to understand if the limited number of elementary flows selected is able to describe the specificities of each sector.

As regards the application of the simplified EPD programme, it needs to be extensively tested to assess its suitability for SMEs. Today we cannot anticipate that eVerdEE will be actually used for certification purpose, but some experiences already exist of its application as a tool for the adoption of life cycle thinking approach at SMEs.

Finally, the results presented have also highlighted, and confirmed, that consistency among different databases is a key point strongly affecting the results of an LCA study. From this viewpoint, the activities led by the European Platform on LCA, which are aimed at harmonising LCA data, are expected to increase reliability and credibility of LCA results (and thus LCA applications) due to the increase of transparency and consistency.

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Beef of local and global provenance: A comparison in terms of energy, CO_2, scale, and farm management

Schlich E., Hardtert B., and Krause F.
Justus Liebig University, Chair of Process Engineering in Food, elmar.schlich@uni-giessen.de

Keywords: Ecology of Scale, Beef, Energy, Energy balance, Carbon dioxide, Farm management, Intensity

Abstract

Food of local origin is said to be eco-friendly because of the small transport distance required. Food from global provenance, on the other hand, is equated with high energy use. However, the specific energy turnover decreases with increasing size of the transportation unit. German beef delivered from intensive[5] but small-scale farms is burdened with high specific energy turnover (HARDTERT 2008). Argentine beef, which is extensively[6] bred on a large scale and transported by cargo ship and truck, proves to have lower energy turnover and CO_2 release (KRAUSE 2008).

Our data demonstrate the main impact of farm management in Germany. The small transport distance to market cannot compensate for the high energy investment required for intensive stock breeding. Global transport of Argentine beef imposes a much smaller burden on the energy balance than is generally believed by the public.

As expected, our data show a declining relationship between specific energy turnover and business size. A declining relationship with business size is also shown for CO_2 release and for primary energy consumption. Additionally, our data indicate a minimum business size necessary to break even. These results corroborate our theory of “Ecology of Scale” (SCHLICH 2004a, SCHLICH 2004b).

Other potential climate impacts include methane and nitrous oxide emissions. Further investigation is intended as to prove any difference comparing extensive and intensive farm management. A shift in paradigm from intensive to extensive farm management in Germany would be advantageous in terms of energy use and carbon dioxide emissions per kilogram of ready-to-cook beef. However, extensive cattle breeding in Germany would face severe disadvantages regarding land use and would significantly reduce German self-sufficiency in beef production.

Introduction

Politically, the apparent advantages of regionally sourced products are currently being articulately emphasized. At the consumer level, regional products are also favorably associated with eco-friendliness. The food supply process is basically determined by a chain-like sequence of interacting steps. The major steps of food supply chains include production and transportation of raw material, transportation of intermediate products (in some cases), processing and apportionment to wholesalers, and distribution for retail sale. Fig. 1 shows the principal components of the food supply chain.

[5] The word “intensive” in this context refers to farming methods that use maximal inputs such as fattening by concentrate feed, as opposed to “extensive”.

[6] The word "extensive" in this context refers to farming methods that use minimal inputs across large land areas, as opposed to "intensive".

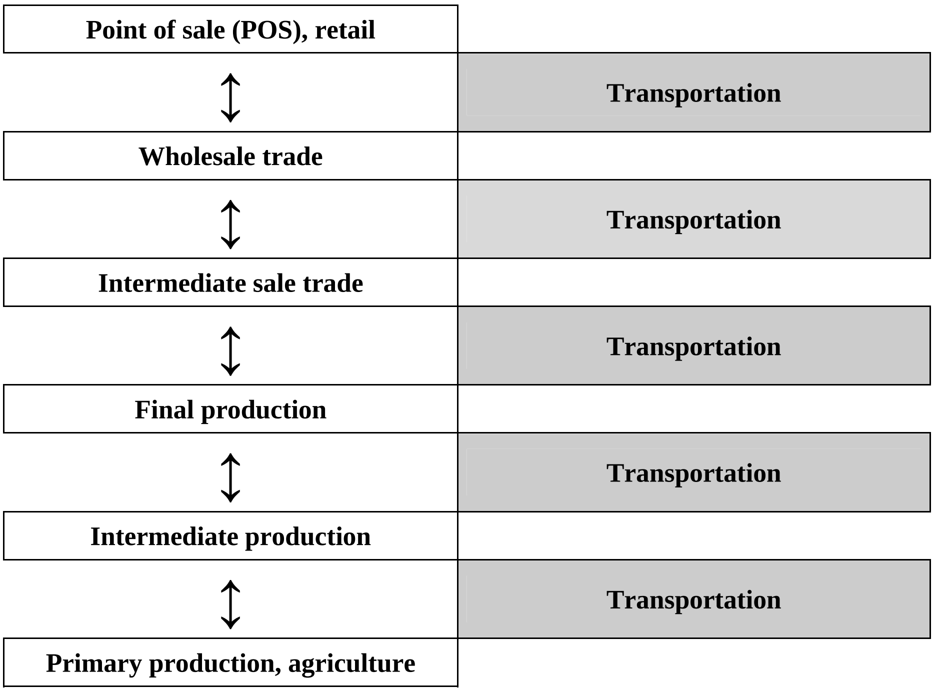

Fig. 1: Supply chain for food

From this figure, it can be seen that transport operations are of key importance in the food supply process. Previous studies focusing on lamb and fruit juice have shown that blanket estimations of energetic favorability solely based on the transport time and transport distance comparing regional and global food supply may not apply to different commodities. Rather, the current state-of-the-art in food supply research emphasizes the explicit influence of scale in conjunction with the eco-friendliness of the food supply chain (FLEISSNER 2002, SCHLICH 2008, SCHRÖDER 2007). In the present work, the term "scale of the supply chain" describes the annual mass throughput of the businesses under comparison (here, the cattle breeders). The investigated interrelationship has been described as "Ecology of Scale" by SCHLICH (2004a, 2004b).

Pork, beef, and lamb are economically relevant to the German meat market. Analyses of the energy dynamics of lamb and pork production have been published by FLEISSNER (2002) and HARDTERT (2008). The energy dynamics of beef production have been investigated by HARDTERT (2008) and KRAUSE (2008) in their doctoral theses. A review of previous scientific publications regarding this topic has been presented there.

Focusing on the fact that the global food supply commonly operates at larger scales than the regional food supply, the central objective of our research is to determine whether the regional food supply shows significant differences in energy turnover in comparison to the global food supply. Additionally, the different conditions of production in distinct countries must be considered. Under certain conditions, favorable initial circumstances may compensate for the environmental impact of long distance transportation, as discussed by JUNGBLUTH (2000) and SCHLICH (2004a).

Method / Approach

We evaluated energy turnover by analyzing food supply chains. Due to EU ordinance 178 (2002), and to other European and national guidelines regarding food traceability, food supply chains can be characterized by tracking all supplier-client relations step-by-step, beginning at the point of sale. Each actor in a food supply chain must document the previous and subsequent steps in the chain, in accordance with legal requirements.

The empirical portion of this study was based upon direct analysis and questioning of breeders, cattle processing businesses, beef trading businesses and freight forwarding businesses. We documented the entire energy turnover of the supply of beef from Argentina by collecting data in situ during a five-week expedition, as well as by e-mail and telephone. Without exception, the analyzed companies of the global food supply form part of one supply chain and have trade relations with each other. We referred to HARDTERT (2008) for data concerning the local beef supply. The energy balance of the local and global supply was based on VDI 4600:1997, Cumulative Energy Demand.

The first step of our research was to define the framework and the boundary conditions of the investigated systems. Previous investigations have shown that calculating the balance of delivered energy according to VDI 4600:1997 provides data of high validity. Based on these calculations of delivered energy, the balance of primary energy and specific carbon dioxide release was calculated by means of the conversion efficiency of alternative fuel sources, i.e., crude oil vs. heating oil, coal or nuclear energy vs. electricity, and natural gas vs. long-distance or local heating. The calculation of specific carbon dioxide release must consider differences in the mix of energy sources involved in the production of electricity or long-distance heating. This requires conversion factors that depend on local, regional or national conditions.

In addition to carbon dioxide (CO_2), beef production also results in the emission of gases like methane (CH_4) and nitrous oxide (N_2O) that are known to contribute to the greenhouse effect. Methane is released mainly into the atmosphere as result of the digestion of fodder by the ruminant metabolism. Additionally the treatment of manure is causing release of either CH_4 or CO_2 depending on anaerobic or aerobic regime of the degradation of organic compounds. Nitrous oxide derives from the fertilization of farmlands with artificial or natural manure, containing nitrogen that is chemically bound as ammonium (NH_4^+), nitrate (NO_3), ammonium nitrate (NH_4NO_3), and urea (CH_4N_2O). Soil bacteria convert part of the nitrogen content of fertilizers into nitrous oxide. It is well known that intensive farming causes more nitrogen compounds to be present in the liquid and solid manure because of the high protein content of the concentrated feed that is used. As a result, more nitrous oxide is released by intensively managed livestock farming than by extensively managed livestock farming. Artificial fertilization and the input of manure to grazing land in order to intensify fodder production are quite common in Germany but unusual in Argentina. Further research is intended as to find out any difference between German and Argentine cattle in CH_4 and N_2O production per kilogram of ready-to-cook beef. In that context the question of different land use by intensive and extensive farm management will be a matter of research as well.

Module

In this context, the term “module” is understood to mean a discrete set of tasks or an operational unit within a supply chain. Hence, "module" differs from the terms “company” and “enterprise”, which are also used. Our definition of the term “module” and presentation of results based on this unit are required in order to clarify the important role of the different structures within enterprises, which are part of entire supply chains.

The differentiation and specialization of the actors that participate in food supply chains increase with the size of the business. Some supply chains are characterized by only few actors. For example, the local production and transportation of cattle feed, breeding of cattle, and distribution of the cattle to butchers are quite often managed by a single entity. In the case of larger structures involved in the European or global beef supply, a great number of companies participate. These companies are individually responsible for breeding, slaughter, various transportation phases, sea transportation, seaport logistics, distribution logistics, wholesale, retail and so on. The term “module” takes these differentiations of function into account.

Functional unit

Beef imported for sale in Germany is limited to meat without tendons and bones. This kind of meat is called “ready-to-cook beef”. In order to compare different supply chains and origins of beef, one kilogram of “ready-to-cook beef” is defined here as the functional unit. This definition respects the point of view of the final consumer at the point of sale, who faces beef of different origins at retailers. Most consumers compare different varieties of beef by taking several aspects of quality into consideration, including the price of 1 kg, sensory characteristics, mad cow disease, geographic origin, breed or farm management. However, the final decision of the consumer in all cases concerns “ready-to-cook beef”, which is delivered to the point of sale by supply chains of different kinds.

In accordance with VDI 4600:1997, specific energy turnover is presented in kWh per kg of "ready-to-cook beef". Carbon dioxide release is presented as specific emission in kg CO_2 per kg "ready-to-cook beef".

Results

The results of our case studies for global and local beef are presented in Tab. 1.

Tab. 1: Overview of results (Hardtert 2008, Krause 2008)

	Delivered energy in kWh per kg	Primary energy in kWh per kg	Carbon dioxide release in kg CO_2 per kg	Beef production in kg per year
Global	2.94	3.87	0.83	94,600
Local1	7.27	9.19	2.11	8,352
Local2	7.94	9.54	2.20	2,885
Local3	5.78	6.67	1.53	18,280
Local4	6.14	7.98	1.83	15,408
Local5	4.66	5.71	1.30	16,196

The relationship of specific energy turnover and specific carbon dioxide release to business size is presented in Fig. 2, 3 and 4.

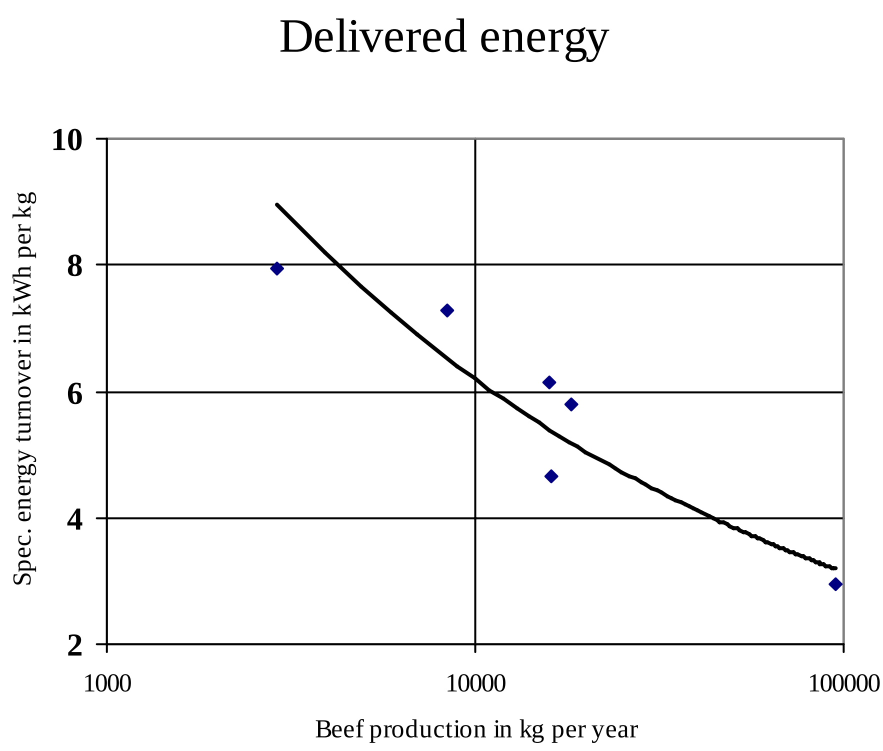

Fig. 2: Specific turnover of delivered energy versus beef production (Hardtert 2008, Krause 2008)

The interpolation of these data by standardized regression analysis is given by equation 1:

$$y = 93.943\, x^{-0.2952} \text{ and } R^2 = 0.87 \qquad \text{(eq. 1)}$$

Primary energy

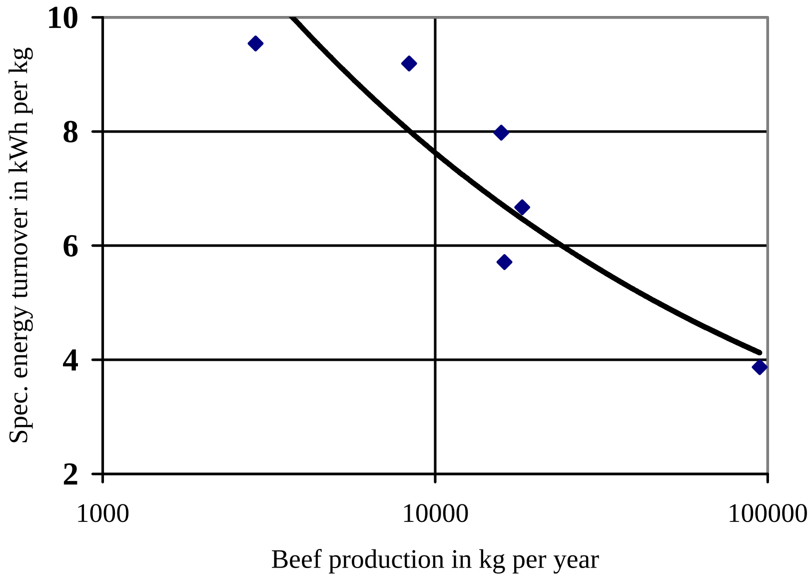

Fig. 3: Specific turnover of primary energy versus beef production (Hardtert 2008, Krause 2008)

The interpolation of these data by standardized regression analysis is given by equation 2:

$$y = 95.164\, x^{-0.274} \text{ and } R^2 = 0.84 \qquad \text{(eq. 2)}$$

Carbon dioxide

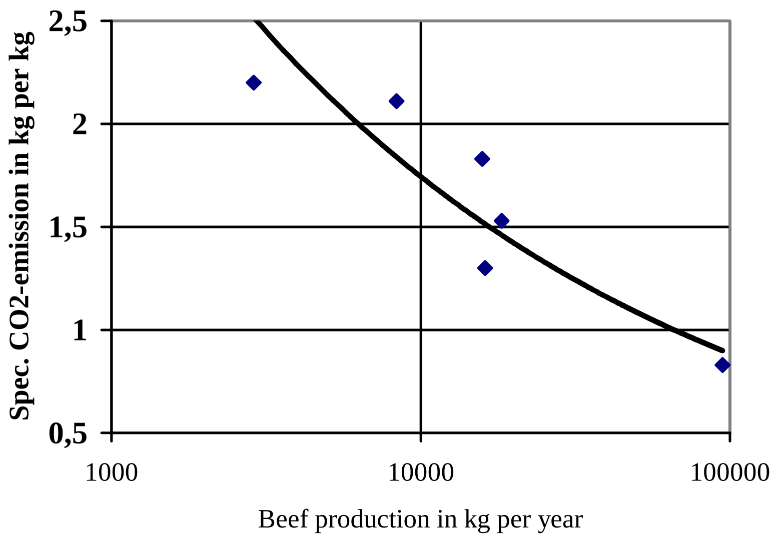

Fig. 4: Specific emission of CO2 versus beef production (Hardtert 2008, Krause 2008)

The interpolation of these data by standardized regression analysis is given by equation 3:

$$y = 26.325\, x^{-0.2947} \text{ and } R^2 = 0.85 \qquad \text{(eq. 3)}$$

Examining the allocation of different energy carriers to the various modules, we find significant contrasts between beef of global and of local origin. "Ready-to-cook beef" originating from Argentina is characterized by long-distance transportation. The energy carriers needed for global transportation – heavy oil and gasoline – dominate the specific energy turnover and, therefore, the specific carbon dioxide release as well. On the other hand, breeding and fattening of cattle in Argentina does not require much energy input because of the extensive farm management that is applied there. Cowboys on horses herd the cattle in Argentina's wide grassland plains. The breed of cattle used in this extensive system requires two years before it is traded and shipped to the slaughterhouses. From there, standard 20-foot shipping containers fully loaded with chilled meat are sent to the Argentine seaport bound for overseas (e.g. Europe, the United States, and Japan). Altogether, the fossil fuels needed for global transport account for about 65% of delivered energy, about 55% of primary energy, and about 62% of CO_2 release (KRAUSE 2008).

At the local scale, the proportions of energy turnover for the different components of the beef supply chain are absolutely different from those at the global scale. Local transportation of “ready-to-cook beef” accounts for less than 10% of delivered energy, primary energy, and carbon dioxide release. In Germany, the inputs required for breeding and fattening cattle dominate the specific turnover of energy. In fact, 72-85% of the entire energy turnover for German beef is needed for the intensive farm management. These numbers include the production and transportation of concentrated feed, which is almost always required in order to breed and fatten cattle in Germany (Hardtert 2008).

Discussion and conclusions

First of all, our data demonstrate the dependency of specific energy turnover on the business scale. As can be clearly seen in Fig. 2-4, the specific energy turnover and specific carbon dioxide release significantly decline with increasing scale of yearly beef production. Some lobbyists might argue that German cattle farms should be examined differently because of the entirely different type of farm management in Argentina. However, our findings definitely point to a declining relationship even if we only consider the data of intensive breeding farms. The calculated coefficients of determination are rather high (84-87%), although the number of cases investigated here is low. The analyzed functions of the regression analyses are hyperbolic. Hence, these data again support our theory of "Ecology of Scale".

Secondly, our data suggest a minimum business size for intensive cattle breeding on German farms, which is estimated to be greater than 50,000 kilograms of "ready-to-cook beef" per year. This corresponds to at least 550 head of cattle[7] on a German livestock farm of reasonable size. Most interesting is the fact that extensive but global beef production without any use of concentrated feed burdens the energy balance much less than intensive livestock farming in Germany, despite the energy costs of long-distance global transportation. The high inputs of energy for intensive farming in Germany cannot be compensated for by long distance transportation of global beef, unless the livestock farming in Germany reaches the minimum farm size mentioned above.

A further comparison of intensive and extensive cattle breeding may illustrate the rather high impact of beef production and consumption in industrial countries like Germany. If the livestock-farming paradigm in Germany were to change from intensive to extensive management, two effects would have to be considered. Firstly, the specific energy turnover and specific carbon dioxide release of German beef would drop to only 20-30% of the values presented here because the inputs necessary for intensive farming and for supplying concentrated feed would be cancelled. Even if we take into account some additional energy requirements for extensive herd management, this type of farm management still turns out to be energy saving. On the other hand, Germany's self-sufficiency for beef, which is greater than 100%, would break down in the case of a shift from intensive to exclusively extensive farm management in Germany.

[7] One head of cattle ready to slaughter weighs about 380 kg alive. Beef at the point of sale (ready to cook, without tendons and bones) corresponds to 48% of that weight, or about 182.4 kg. Breeding and fattening of the cattle takes two years. Hence, a reasonable farm size is calculated by $>50{,}000$ kg/182.4 kg $\cdot 2 > 548.25$ head of cattle.

Finally, both nutritional and environmental considerations suggest that a general reduction of beef consumption may be desirable. The recent promotion of exclusively "regional" beef in Germany – by, for example, regional politicians, farmers' associations and beef-based fast food restaurants – does not solve the pressing challenges created by the high-meat-consuming lifestyle in industrial countries like Germany.

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An analysis of the present food's transport model based on a case study carried out in Spain

Aranda A.[1], Scarpellini S., Zabalza I., Valero Capilla A.
[1]CIRCE -Centre of Research for Energy Resources and Consumption, University of Zaragoza, Zaragoza, Spain, alaranda@unizar.es

Keywords: Local Food; Sustainability; Food; Ecoefficiency; Ecolabelling; Transport, Spain

Abstract

The activity of the food sector is strongly linked to the behaviour of the consumer. Historically, in Spain the 33,000 companies that make up the food sector represent around 8% of the final energy consumption. In Spain the tendency towards globalisation in the sector, together with a sharp increase in the import and export of finished or semi-finished products, and the increase in consumption of ready-made and frozen products means that the energy invested in products is higher and the environmental impact greater.

The paper analyses the current trends of the food sector in Spain with the historical record. The current and future degree of sustainability is analysed using Life Cycle Assessment (LCA). The shopping basket of the average Spanish citizen has been used in the Regional capital city of Aragón, Zaragoza, as a functional unit, which is compared with the situation that would be obtained with a higher level of consumption of local food, or if packaging were reduced or even if another type of packaging were used.

The ecolabelling of food products may be a discriminatory sign against certain less desirable products. The results presented in this paper may serve to orient energy efficiency or greenhouse emissions policies for the Spanish food sector.

Introduction

The food sector has fundamental importance, as it is one of the sectors with the greatest impact on ecosystems, from problems of soil erosion and depletion and contamination of subterranean water in agricultural areas to greenhouse emissions in processing and transport. In addition, it is a sector that is strongly linked to the behaviour of private consumption. Moreover, the low cost of transport has reinforced the specialisation, centralisation and globalisation of the food market, increasing both the volume of goods transported and the distance travelled (van Veen-Groot & Nijkamp, 1999).

The environmental cost of food production and transport is not included in product price. It is often cheaper to import a product from far away than to produce it locally. Thus a growing trend has been observed in the distance travelled by foods, known as "food miles" (Smith *et al.* 2005). Concrete data collected in the United Kingdom found that the annual quantity of foods transported by road in HGVs since 1978 has increased by 23% and the average distance of each journey has increased by 50%.

From the perspective of emissions, it has emerged from studies, such as the one carried out by Edwars-Jones *et al.* (2008), basing the benefits of local consumption on a reduction in the energy consumed by transport can be shown to be insufficient, given the significant impact brought about by the production and manufacturing stages. In this paper the impact of food transport has been analysed from a global point of view, independent of the place where the environmental impact improvement could be obtained.

At the same time, the lack of an established calculation method and limits to the system defined when applying the LCA to the foods have been detected. The emissions result obtained through the analysis will depend on both of these (Buckwell, 2005) and this will permit comparison and extrapolation between the results of different studies. As an example of the need to standardise criteria when establishing the system limits and the calculation method, studies can be found that conclude that the energy consumption in importing apples produced in New Zealand to Sweden is greater than if they were grown directly in Sweden (Stading, 1997), while others suggest otherwise (Saunders, et al 2006).

Numerous studies have to be carried out for each country. It is due to the considerable differences to be appreciated between various geographic areas and the importance of climatology in this kind of analysis. For instance, the Carlsson-Kanyama & Linden 2001 studies demonstrated the improvement of the environmental impact in Sweden caused by the increment of meat consumption and taking into account a global perspective of foodstuff delivery.

Some other studies demonstrate the benefits of local production of vegetables in lands closed to urban areas, depending on the density of the population. For example Ghosh, S. *et al*. (2008) analysed the potential local vegetable production in the region of Auckland, New Zealand, and he calculated the environmental improvement obtained using the local production of vegetables.

Up to now, various LCA studies reveal that local production is more energy efficient than non-local, due primarily to the savings in transport. Nevertheless, this assertion must not be extended to all products nor to all areas, as the LCA includes numerous variables that are difficult to characterise according to location, type of crop, season, etc., making a specific analysis necessary for each product. An example of this is data produced by analysis such as that of Smith *et al* (2005), which determines that the energy efficiency of tomatoes imported from Spain to the United Kingdom is less than that of producing them locally in greenhouses. This, in turn, is evidence that the different production systems are determining factors in the identification of the most environmentally favourable option. Contrarily, estimating the greenhouse emissions due to transport is relatively simple, given that the emissions ratios used are widely known.

Approach

The energy efficiency measures considered in the policies developed by different countries in the agricultural and food sector are fundamentally for saving energy in processes: management of steam and condenser systems, installation of heat pumps, speed variators in engines, compressors, substitution for natural gas, improvements in lighting and electricity, improvements in boilers and heat exchangers, heat recovery from process fluids, improvements in cooling stations, improvements in dryers, assessment of waste, recycling and recirculation, etc. The implementation of the Spanish Energy Efficiency Strategy (E4) adopted by the Spanish Ministry of Industry, Tourism and Trade is the instrument used by the Spanish Government at present in order to achieve the goals for energy efficiency in Spain.

At present, the energy audits in the sector are well developed and have shown, generally speaking, good results in countries such as Australia, Belgium, Finland, France, Germany, Ireland, Netherlands, Portugal, Turkey, United Kingdom, Spain and Denmark (a country that follows the "green tax" scheme that reduces taxes for companies that apply the measures of the energy audits carried out). Another tool is voluntary commitment from companies to emission-reducing actions. For example, in the Netherlands around 1,200 industrial companies are included accounting for 90% of primary energy consumption. Canada and Norway are also good examples.

Nevertheless, measures aimed at reducing the energy consumed transporting the end products are not common. This paper aims to analyse the importance of energy consumption in food transport and the advantages of promoting a more local purchasing system, and to evaluate the system for containers and packaging applied to food.

The Observatory of Consumption and Food Distribution (Observatorio del Consumo y la Distribución Alimentaria) of the Spanish Ministry of the Environment and Rural and Marine Affairs, MAPA (2007) goes deeper into the analysis of consumption habits and food distribution highlighting facts such as:

- When buying fruit, greens, fresh vegetables and other foods, the factors we consider most are the quality and appearance of the product, and least, the ecological production and the country/area of origin. Some 84% of the population do not recognise the specific labelling of ecological products, although 27.5% maintain they have consumed these products at some point. It is necessary to promote the eco-label where food origin is specified in order to inform the consumer adequately when buying. In establishments selling food, the consumer turns more and more to packaged products, smaller formats and ready meals.

Method

Due to its varied origin, long distance food transport has almost doubled between 1968 and 1998 (Jones & Hird, 2001). International food trade has increased by 184%, while food production has only grown by 84%, which means that food travels more and further by land, sea and air. Specifically for Spain, in the year 2007, the quantity of this exported was 35% of all food exported during the period 1995-2004, with a similar figure for imports. In total almost 50 million tonnes of foods crossed Spain's borders, whether entering or leaving the country, confirming the exponential trend in food transport in recent years.

Regarding the list of the foods most consumed in Spain, the most significant are gathered in Tab. 1:

Tab. 1: List of the most consumed products in Spanish shopping baskets. Source: Spanish National Statistics Institute, 2006

	Total quantity consumed	**Average quantity consumed per person per year**
Eggs (units)	4,663,386,000	108.2
Mineral water (l)	2,373,506	55.1
Milk (l)	3,857,778	54.8
Bread (units)	1,953,370,000	45.3
Fizzy drinks (l)	1,535,932	35.6
Potatoes (kg)	1,280,820	29.7
Citrus fruits (fresh/refrigerated) (kg)	1,001,782	23.2
Fresh fruit (kg)	870,061	20.2
Beer (l)	755,753	17.5
Fruit juices (l)	733,527	17
Yoghurts (kg)	717,750	16.6

With the objective of extrapolating to a national level, a city, Zaragoza (Spain), inhabited by 750,000 people, and a shopping basket identical to the one used in the calculation of the consumer price index (CPI) were selected. An energy study was carried out in two scenarios: food consumption according to the current import-export trends and the consumption that would be obtained if, though a system of ecolabelling, ecotaxes, consumer information, awareness-raising and voluntary agreements with wholesalers and retailers, part of this food consumed were locally produced, eliminating long distance transport.

The Spanish data (obtained from the Spanish Ministry of Industry, Tourism and Trade) shows the importance and the increasing evolution of imported and exported foodstuff. In 2007, exported foods corresponded to 35% of all exported food during the period 1995-2004. A similar situation was detected for imports. Almost 50 millions tonnes of foodstuff in total crossed Spanish borders during 2007 (24.8 t went out and 24.3 t come in)

Despite the importance of the whole life cycle of a food, given the absence of a consensus for establishing limits in the system studied and the lack of databases that allow us to calculate the impact due to the life cycle of each food in specific geographical zones, in conjunction with the substantial burden that transport generally has on the global impact of a product, hereafter the study centred on estimating the emissions avoided due to transport, supposing the existence of a sufficient local market, for those products that can be produced locally.

Numerous studies have been carried out to analyse vegetable, fruit and other foodstuffs in Spain where suggestions for environmental impact improvement are clear: orange production on the eastern coast (Sanjuán, N. et al, 2005); wine production (Aranda, A. et al, 2005); tomato production (Antón, A. *et al*., 2005) or meat production (Nuñez, Y., 2005), etc. However, in almost all of them, food transport and distribution are not taken into account in the LCA analysis, which is the principal objective of this paper.

At this point we must highlight the importance of the efficiency of the transport. In general, the most widely used long distance transport system is the HGV (40t) between the supplier and the distribution centres, which generally optimises the loads and routes, reducing the impact per tonne transported. However, with more local suppliers, the reduction of emissions due to the shorter distance travelled can be counteracted by the use of smaller vehicles and/or not making full use of the carrying capacity.

To assess the energy cost of food transport, the limits of the system were defined to include the energy and material costs in the manufacture of the vehicles, and those arising from their use (fuel consumption), maintenance (tyres, oil, etc.). Similarly, the contribution to the impact of the regional distribution of the fuel used in the transport has been considered. The impact of the construction of the infrastructure necessary for this transport was not taken into account as it is not exclusive to goods transport, but is also used for mobility. If there were a change in the mode of transport, it is true that the existing infrastructure would not be necessary, nor would that envisaged for the future, but this is very difficult to quantify. The parameters used, obtained from the Ecoinvent database, for the different types of transport are the following (Tab. 2):

Tab. 2: Energy consumption and emissions derived from the use of different means of transport

	Energy (kcal/tkm)	Emissions (gCO_2eq/tkm)	Eco-indicator 99 (mPt)
40t lorry	654	164	15.1
28t lorry	867	221	20.8
16t lorry	1,209	315	29.6
Transoceanic freight ship	35	10.5	1.3

In assessing the energy cost of food transport, the limits of the system were defined to include the energy and material costs in the manufacture of the vehicles, and those arising from their use (fuel consumption), maintenance (tyres, oil, etc.).

In general, the most widely used long distance transport system is the HGV (40t) between the supplier and the distribution centres, which generally optimises the loads and routes. In the case of local distribution from local production centres a 28 ton lorry is commonly used in spite of it being the most inefficient form from an energy point of view.

In a comparative of the different means of transport considered, assessed in terms of environmental impact according to Eco-indicator 99 H/A (Goedkoop, & Spriensmaa, 2000), Tab. 2 shows how this indicator perfectly models the energy costs and, to a lesser degree although with very acceptable approximations, the environmental cost arising from greenhouse emissions.

To study the influence of transport in the food sector, for example, we analysed the origin of the products sold in Saragossa. We must point out that the specialisation of the crops in certain geographical zones and of food production and preparation can give rise to reduced energy consumption that compensates for the increases due to transport (Edwars-Jones *et al.* 2008).

This LCA presents a certain degree of simplification. The use of commercial databases to complete the inventories of each of the raw materials and processes or the application of impact assessment procedures already implies a simplification in itself. The LCA method was limited to the analysis of the transport energy consumption and the carbon footprint generated.

Results

For the study, the shopping basket was broken down into **"fruit and vegetables"** and **"other products"**. For the former, according to the latest data published by the wholesale market selling in Saragossa (Mercazaragoza), 2005 saw 172,155 tonnes of fruit and vegetables sold. Of these, 91,566 t were fruit, of which only 1% were sold directly through farmers from the surrounding area. A similar situation occurred with the 80,589 t of vegetables, of which only around 6% were sold by market gardeners from the area. The specific data shows the following sales distribution (Tab. 3):

Tab. 3: Amounts consumed and primary origin of the main fruit and vegetables. Source: Mercazaragoza

VEGETABLE	QUANTITY (t)	ORIGIN		FRUIT	QUANTITY (t)	ORIGIN
Potato	18,100	Almería		**Banana**	11,158	Canary Islands
Tomato	12,580	Almería and Holland		**Orange**	29,075	Valencia and Brazil
Lettuce	8,705	Murcia		**Melon**	6,563	Murcia and Castile-La Mancha
Onion	6,269	Aragon		**Watermelon**	5,522	Almería and Valencia
Lettuce	5,041	Almería		**Apple**	3,782	Italy and Lérida
Green bean	3,967	Morocco		**Tangerine**	3,710	Valencia
Borage	2,477	Aragon		**Long stem strawberry**	3,341	Almería and Morocco
Pepper	2,430	Almería		**Peach**	3,167	Aragón and Lérida
				Lemon	2,601	Valencia
				Pear	2,564	Belgium, Lérida and Aragon

Taking into account these amounts and the distance they travel for sale and using the data of the different modes of transport used (Tab. 2), it is possible to estimate the energy consumed in the transport of this food. Moreover, considering the existence of a local market (radius of less than 200 km) capable of supplying the market with the products that can be produced due to the climate, and taking into account that in this case the transport would require lighter modes of transport (28t instead of 40t), the energy savings resulting from the change to a more local consumption can be estimated (Tab. 4).

Tab. 4: Comparative of energy consumption and emissions of fruit and vegetable consumption at present and in a local market

CURRENT MARKET	toe	t CO_2 eq	LOCAL MARKET	toe	t CO_2 eq
Potato	917	2,307	**Potato**	314	802
Tomato	971	2,440	**Tomato**	218	557
Lettuce	321	806	**Lettuce**	151	386
Onion	109	278	**Onion**	109	278
Iceberg lettuce	256	642	**Iceberg lettuce**	87	223
Green bean	305	635	**Green bean**	69	176
Borage	60	110	**Borage**	43	110
Green pepper	123	309	**Green pepper**	42	250
	3,062	**7,529**		**1,033**	**2,781**
ENERGY SAVED: 2,029 toe					
EMISSIONS AVOIDED: 4,748 tCO_2eq					

CURRENT MARKET	toe	t CO_2 eq	LOCAL MARKET	toe	t CO_2 eq
Orange	1,573	4,087	**Orange**	750	1,910
Banana	760	1,935	**Banana**	760	1,935
Melon	239	600	**Melon**	114	291
Watermelon	108	272	**Watermelon**	96	244
Apple	527	661	**Apple**	65	167
Tangerine	97	244	**Tangerine**	97	244
Long stem strawberry	169	425	**Long stem strawberry**	169	425
Peach	31	104	**Peach**	55	140
Lemon	68	171	**Lemon**	68	171
Pear	129	245	**Pear**	44	113
	3,702	**8,745**		**1,469**	**3,707**
ENERGY SAVED: 2,232 toe					
EMISSIONS AVOIDED: 5,038 tCO_2eq					

For this analysis we considered that, with the exception of citrus fruits, bananas and strawberries, which would not be possible to grow locally as the climate is unsuitable, for this reason the same energy consumption was considered in both scenarios. The rest of the products could be supplied from local crops.

The inclusion of some foods, as for example, watermelons and peaches significantly affect the average as they disproportionally use more energy to get them to the consumers (40 t lorry instead 28 tonnes lorries used in the local transport).

Looking at the data, and taking into account the nutritional values of these products, many of them need several times more energy to be transported than the energy they are capable of supplying as food, demonstrating the inefficiency of the current food distribution systems (Tab. 5).

Tab. 5: Energy for transport vs. the nutritional value of different kind of vegetables.

Vegetable	Energy for transport (kcal/kg)	Nutritional value (kcal/kg)	Energy for transport / Nutritional value
Potato	507	860	59%
Tomato	772	210	**368%**
Lettuce	368	130	**284%**
Onion	173	380	46%
Iceberg Lettuce	507	110	**461%**
Green Bean	768	286	**269%**
Borage	241	138	**175%**
Green Pepper	507	195	**260%**

In this analysis the seasonal availability of all the different kinds of vegetables has not been taken into account. The objective is to calculate the impact of transport for foodstuff separately. A detailed study for each vegetable would be necessary in order to introduce this factor into the specific analysis.

Regarding the **rest of the most common products** in the shopping basket, to achieve a representative sample, the data published by the Spanish National Statistics Institute (Instituto Nacional de Estadística), that determines which products families consume and which 57 items make up the shopping basket for calculating the CPI, has been taken. Their origin has been studied in different commercial establishments to analyse the impact transporting them causes.

Insufficient clear information on the geographic origin of the food we consume was observed on the labels. In most cases, it only specifies the product was packaged and produced in Spain. We also established that each product is offered by a large number of different brands, coming from at least 300 km away, without considering the origin of the raw materials with which it was made, which is impossible to know in most cases.

The products analysed and their relative distances from their origin (an average distance of the different places was obtained) are the following (Tab. 6): (the % indicates the quantity of products sold in a local market).

Tab. 6: Products analysed and their consumption and origin

	ORIGIN	ANNUAL CONSUMPTION PER PERSON
Eggs (kg)	Saragossa 33% - Guadalajara 260km	7.6
Water (l=kg)	Aragon 46% - remainder average 380km	55.1
Milk (l=kg)	Aragon 17% - remainder average 610km	89.5
Juices (l=kg)	Aragon 0% - remainder average 450km	17
Chicken (kg)	Aragon 0% - remainder average 365km	13.4
Olive oil (l)	Aragon 36% - remainder average 720km	11.61
Pulses (kg)	Spain 16% - remainder average 5000km	3.8
Rice (kg)	Aragon 0% - remainder average 463km	6
Pasta (kg)	Aragon 20% - remainder average 704km	5.1

In the "Current Market" scenario the food transport element has been considered as having been done by road with a 40t lorry while for the "Local Market" a 28 ton lorry has been selected due to the shorter distances from the production areas.

It was supposed that imports have come to the peninsula by ship. Assuming that all these products, which are the most consumed, could be produced locally, the different energy consumptions estimated per person for a typical annual basket are summarised in Tab. 7, showing energy savings in transport of 28,708 kcal/year (42% of the consumption in transport):

Tab. 7: Comparative between the market at present and the local market of the most sold products (not including fruit and vegetables). Data per person per year

	CURRENT MARKET		LOCAL MARKET	
	kcal/year	*kg $CO_2eq/year$*	*kcal/year*	*kg CO_2 eq/year*
Eggs	1,424	305	1,314	335
Water	11,790	2,980	9,558	2,441
Milk	32,273	7,949	15,525	3,965
Juices	5,003	1,257	2,949	753
Chicken	3,198	633	2,324	594
Olive oil	4,223	1,045	2,014	514
Pulses	2,722	387	659	168
Rice	1,817	456	1,041	266
Pasta	2,525	506	885	226
	64,976	**15,519**	**36,268**	**9,264**

Also according to the data published by the Spanish National Statistics Institute, the annual quantity of fruit and vegetables consumed is 70.8 and 74.9 kg respectively, so with the results obtained previously for these two food groups, those obtained considering the former basic basket provide the following results (Tab. 8). Here, the energy consumption of both options is determined per person.

Tab. 8: Comparative between the market at present and the local market of the most sold products. Data per person per year

	Energy (kcal/person·year)		EMISSIONS $gCO_2eq./person·year$	
	CURRENT MARKET	LOCAL MARKET	CURRENT MARKET	LOCAL MARKET
Other products	64,976	36,268	15,519	9,264
Fruit and vegetables	75,173	27,548	18,129	7,169
	140,149	**63,816**	**33,648**	**16,433**
	ENERGY SAVED 76,333 *kcal/person·year*		**EMISSIONS AVOIDED** 16,433 *$gCO_2eq./person·year$*	

Conclusion

If this model of local food consumption compared to the current model of the use of import-export is extrapolated to the whole of Spain, with 44 million inhabitants with similar dietary habits, the global energy savings (due to the LCA of the fuel and the LCA of the vehicle) would be of the order of 340 ktoe per year. The measures considered in the Spanish policy for energy efficiency, the "Spanish Energy Efficiency Strategy (E4)", in the food, drinks and tobacco sector imply savings of 414 ktoe per year, so the promotion of the consumption of local food and drinks can achieve results that quantitatively compliment those of the E4. Regarding the emissions avoided (principally due to fuel consumption), these would come to around 720 $ktCO_2eq$.

In order to extrapolate the results to transport impact of the food sector in Spain, it is necessary to take into account that not all foodstuff have been analysed, the subject of this study was the principal foods consumed, fruit and vegetables. To determine the value of these figures, final energy consumption of the sector in Spain in the year 2008 is 3,150 ktoe/year (8% of the whole industrial sector). If the tendency will not change, the energy consumption in Spain in 2012 will be 42.1% more than in 2000, this being absolutely unsustainable. The principal problem is that no actions for the reduction of energy consumption in goods transport have been scheduled at present in Spain. It would represent and important goal for energy saving and emissions reduction if this were to be done.

If the environmental impact of transport were to be taken into account adequately and the local production promoted, we could improve the energy saving in the sector up to 80%.

It is important to note that regarding the products studied in the shops, 9 of the 16 types of products selected were not locally represented, so the potential for their introduction is very high. Therefore the energy saving in transport has to be one of the principal actions for an energy efficiency policy.

Ecolabelling could be another boost in its favour provided the production has been obtained and manufactured according to the technical standards of environmentally respectful production and the sustainability and traceability criteria established.

Agricultural activities, as well as stockbreeding and timber exploitation, have represented the natural capital of the territory of Aragon for hundreds of years. The ecosystems are suffering progressive degradation and it is necessary to maintain traditional activities (no intensive exploitation) in rural areas that are more environmentally friendly. The environmental and social costs would be very high if these rural activities were not maintained and the local production of some foods could be a feasible and adequate instrument in order to avoid this problem.

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Greenhouse Gas Assessment of Ben & Jerry's ice-cream: communicating their 'Climate Hoofprint'

T. Garcia-Suarez[1], S. Sim[1], A. Mauser[2] and P. Marshall[2]
[1]Safety and Environmental Assurance Centre, Unilever, MK40 1LQ, Sharnbrook (Beds), UK
[2]Ben & Jerry's UK Pad, 1st Floor, Goswell House, Goswell Hill, Windsor, SL4 1DS, UK
[3]Ben & Jerry's NL Pad. PO Box 160, 3000 AD Rotterdam, The Netherlands
Tirma.Garcia-Suarez@unilever.com.

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Abstract

There is increasing interest in the calculation of carbon footprints (CF) of products for a variety of purposes. However, calculating the CF presents many challenges and concepts like "carbon equivalents" are not readily understood.

This paper describes a greenhouse gas "mass-balance approach" that has been used to calculate the CF of the annual European production and sales of Ben & Jerry's ice cream. The approach allows for the assessment of a whole portfolio of products in a cost and resource efficient manner.

The results of the CF exercise identified hotspots in the life cycle, namely from ingredients (dairy (17%)) and the retail (46%) stages. With this understanding, the business has a programme to achieve reduction in those areas (on top of reduction in its own production process) by researching new refrigeration technologies and partnering with dairy farmers in a sustainable agriculture programme (known as 'Caring Dairy'). Offsetting the remaining GHG emissions has enabled the brand to become 'climate neutral'; this is incorporated in brand communication.

1. Introduction

The greenhouse gas (GHG) emissions associated with a product (goods and services) along its life cycle are often referred to as a "carbon footprint" (CF) (European Commission, 2007). There is an expectation within many Non-Governmental Organisations (NGOs) and businesses that consumers will soon demand products with low CFs (Berry *et al*, 2008; Carbon Trust, 2006); thus, the desire to calculate and communicate the CF of products and services is increasing. However, calculating these footprints presents many methodological challenges. In addition, concepts like "carbon equivalents" are not readily understood by consumers.

Ben & Jerry's (B&J's) manufactures ice cream for the US and European markets. The brand's activities are guided by its 'social mission', comprised of 5 key themes: natural ingredients, sustainable farming, fair partnerships with suppliers, peaceful activism and minimal eco-footprint. Regarding the latter, B&J's have been pursuing initiatives on climate change since 2003. This paper describes an approach for calculating the annualised CF ("Climate Hoofprint") for B&J's ice cream manufactured and sold in Europe. It is the latest refinement of CF calculations which the brand has undertaken to understand GHG contributions across the life cycle of B&J's products. Within Europe, B&J's ice cream is sold in 26 flavours, and in three formats for in- and out-of-home consumption: single serving tubs (150ml), pint tubs (500 ml) containing 4 servings and bulk formats (4.5 l) used in B&J's 'scoop shops'. This number of flavours and formats presents numerous challenges for carbon footprinting: these are discussed below. So too is the use of the results for management and brand communication activities.

2. Method

2.1 Introducing the mass-balance approach

The carbon footprint of products is normally estimated using process-based (standard) LCA; however, this approach was considered impractical for determining the CF of all B&J's ice cream manufactured in Europe. Adherence to a process-based LCA approach would imply estimation of the global warming potential (GWP)[8] of every single stock keeping unit (SKU) manufactured in Europe. This would require up to 46 separate CFs for each market country to include all 26 ice cream flavours ('vanilla', 'vanilla toffee crunch' 'cookie dough' etc) in their different formats. The cost and resource implications are clearly significant. Alternatively, estimating the CF for just one SKU and extrapolating the results to the total European production of all flavours & formats could be overly simplistic. Thus, whilst process-based LCA is appropriate for tactical decision making at the level of individual products, an alternative approach is required if the results are intended to inform strategic management and for communication activities at a brand level.

For this reason, an annualised 'mass-balance' approach was developed. The reference unit considered is the total annual ice cream production at B&J's European factory. From the production volume and using recipes and waste percentages, volumes of raw materials were calculated and traced back to their origins. Similarly, volumes of finished product were tracked through to distribution centres, retail outlets and homes for each market country (Fig. 1). Tracking mass flows in this way makes calculation of the CF of a whole product portfolio possible, whilst retaining a life cycle perspective. In addition, data requirements are aligned with current data management systems within the business (e.g. reporting of utilities at the level of a factory site, where several products are produced in parallel and in batches). This avoids allocation dilemmas that often persist in standard LCA, where data requirements (e.g. energy required for the vanilla ice cream production line) are mismatched with the current site-level data management systems.

2.2 Scope and System boundaries

The annual European production and sales of Ben & Jerry's ice cream in 2006 (all flavours, formats and retail outlets) were taken as the basis for this study: the complexity implied by this scope is summarised in Tab. 1. The fore- and background systems are shown in Fig. 1. The foreground system is defined as that which is most directly affected by B&J's and where specific, albeit site-level, data were likely to be available, i.e. the manufacturing process, dairy farming (there is a partnership between B&J's and the farmers under a scheme called 'Caring Dairy') and the initial distribution step in the supply chain. The background comprises all other ingredient and packaging inputs to manufacture, chilled distribution, storage in different countries and refrigeration in retail and consumer homes. Disposal of factory waste and product packaging were not included because there was insufficient information to describe specific disposal routes. Whilst average European data for the disposal & treatment of paper are available, their use may have been misleading since B&J's pints are coated with PE, possibly restricting options for waste treatment in some markets. Tab. 1 summarises the life cycle and data coverage for this study, as well as the critical choices and assumptions made.

[8] Whilst is it clear that process-based LCA is normally used to estimate a range of environmental impacts including, but not limited to GWP; discussion of the approach in this paper considers utilisation of the approach for carbon footprinting purposes only.

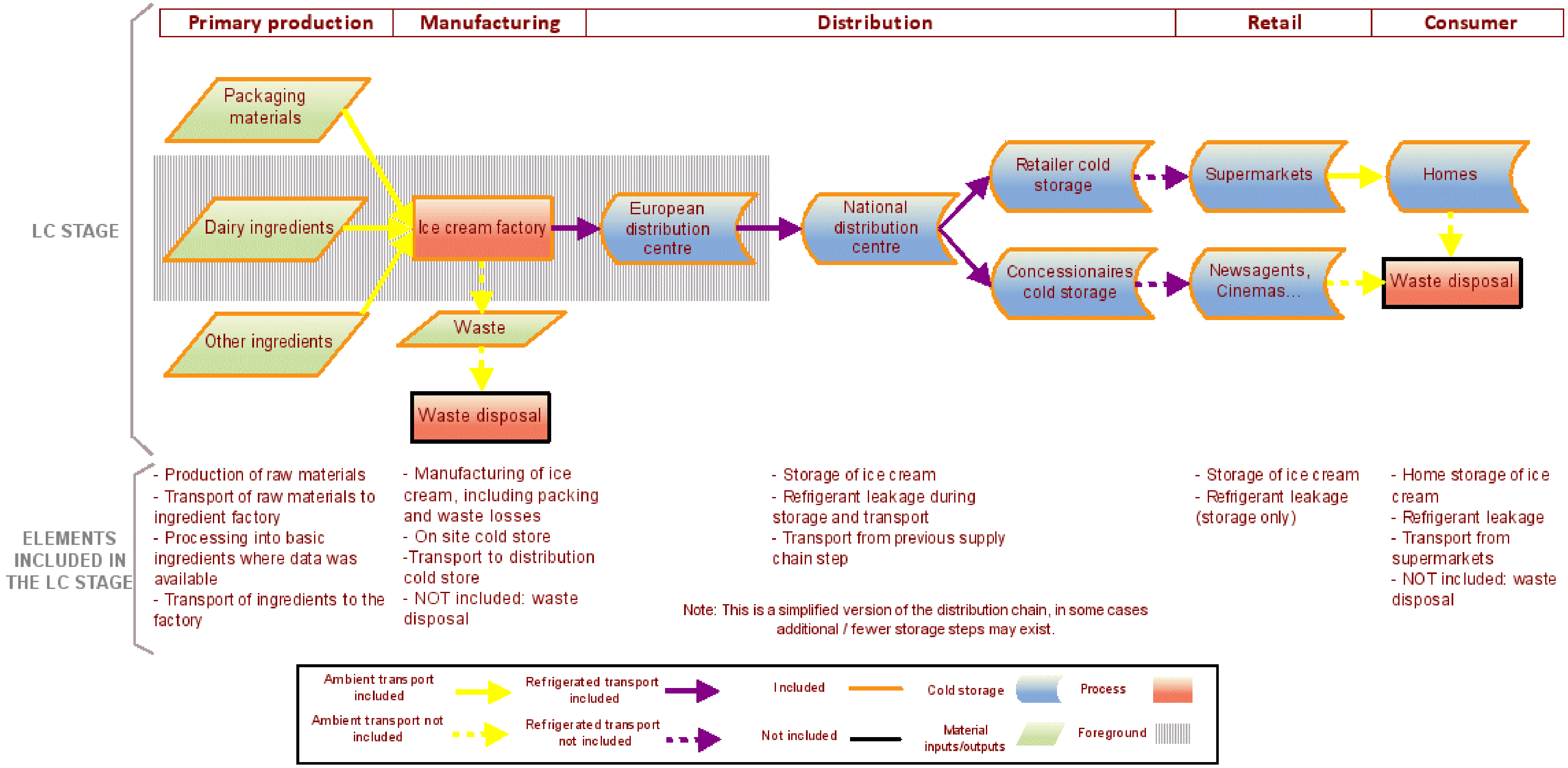

Fig. 1: Process flow diagram of B&J's ice cream: typical flows are included although not specified in the diagram (energy, emissions).

Tab. 1: Summary of Key Study Parameters

Scope of the Study
- Total European production and sales of B&J's ice cream - 2006 - 26 flavours - 3 formats (150 ml 'shorties', 500ml "pint" tubs, 4.5 litre bulk tubs for scoop shops) - 46 SKUs (i.e. flavours in different formats) - 57 ingredient types (i.e. not considering variations such as ground, sliced and chopped almonds) - Ice cream sold in 11 countries in Europe - Joint production (ice cream flavours produced in batches in a factory which also produces other types of ice cream)
Life Cycle Coverage

- Life cycle stages excluded from the study:
    - factory waste disposal
    - packaging waste disposal at end of life
    - transport from retailer cold stores to retail outlets

Data Coverage
- GHG data for 41 ingredients included, covering 98% of the ingredients by mass. The average of the GWP of these was used for the other 2%. - Potential carbon emissions from land use and land use change were not considered since secondary data were mainly used for ingredients and they did not include these emissions. - Where GHG emissions data were lacking for the production of ingredients (e.g. for walnuts), the average for all other ingredients for which data were available was used; this is 1.08 kg CO_2e per kg of ingredient. - Where possible, all production processes for the ice cream ingredients have been considered. However, for some ingredients the CO_2e emissions are likely to be slightly underestimated since the amount of energy associated with certain processing steps is not known - All energy data used in warehouses in the B&J's European supply chain are extrapolated from energy data documented in the Hams Hall carbon audit report prepared by the Carbon Trust (Burrows and Bassett, 2006). Hams Hall is a very modern and highly efficient warehouse; emissions from other warehouses and distribution centres which are older and less efficient may therefore be underestimated.
Data Variability
- Variability in data was considered for ingredients reflecting the range of GHG figures derived from different studies and allocation methods used in those studied (e.g. mass vs. economic). - Data variability in other life cycle stages has not been considered: this is an area for further work, particularly considering seasonal effects on energy requirements for refrigeration and length of storage in retail and consumer homes.
Other Critical choices

- Allocation by mass where performed by the authors (according to the allocation hierarchy in ISO 14041); relevant for ingredients such as egg white and egg yolk.
- Varied allocation methods were used in data from literature.
- Where specific data for distribution of ice cream in certain markets were unavailable, transportation distances for countries where data were available were averaged and applied.
- Green electricity used in manufacturing is generated from hydro-power, which has a minimal effect on the overall GWP results. To simplify communications with the brand this was assumed to have a GWP of zero.
- CO_2 emissions from trucks carrying frozen products were assumed to be 30% higher than for ambient transport and 70% higher for chilled products (UL Lead Engineer Refrigeration, 2008). This accounts for increased energy use and refrigerant leakage.
- On average, the ice cream storage time along the supply chain was distributed as follows:
    - 33% of the time at European distribution centre (DC) and another 33% at national DC
    - 4% in the customers' and concessionaires' cold stores
    - 19% of the time in supermarket freezers or small stores cabinets and scoop shops
    - 11% in home freezers (where relevant, i.e. products purchased in supermarket)
- Refrigerant use and leakage:
    - Ammonia was assumed to be used as a refrigerant in the European DCs, National DCs and 50% of the Customers' cold stores. Ammonia has a GWP of zero
    - R-404A for all other cold stores and frozen transport with a leakage rate of 20% per annum. This refrigerant is also assumed to be used in supermarkets, but with an 18% leakage rate.
- The average stock of supermarket freezer cabinets is assumed to be 60% upright freezers and 40% chest freezers.
- On average 60% of the capacity is utilised in freezer cabinets

2.3 Data Sources

Data related to the foreground were collected from four main sources. For dairy products they were extracted from an in-depth LCA of milk from specific B&J's suppliers. Specific data were available from the farmers that supply B&J's with milk products due to supplier involvement in a sustainable agriculture programme known as "Caring Dairy" (Jansen, 2005). When compared with other dairy LCAs (Thomassen *et al*, 2008; Williams *et al*, 2006; Hospido *et al*, 2003), where both economic and mass allocation have been used, Jansen's results fall in the middle of the range for GWP per kg of fat and protein corrected milk (FCPM).

Data for the manufacturing stage were extracted from Unilever's Environmental Performance Reporting system; these are specific for the B&J's ice cream factory in Europe. Energy data related to cold storage of ice cream in distribution originated from a carbon audit of one of the European distribution centres carried out by the Carbon Trust (Burrows and Bassett, 2006). Transport distances and loading factors were provided by Unilever's regional sales managers. A summary of data sources is presented in Tab. 2. It should be noted that the consumption of ice cream is seasonal. Peak ice cream consumption occurs in the summer months, coinciding with higher ambient temperatures which influence energy requirements for refrigeration in transit, retail and consumer homes. For the purposes of this study, data which represent the annual average energy and material inputs have been used.

Tab. 2: Data sources

Ingredients	**Source**	**Data**
Milk / Cream	Jansen (2005)	GWP
Sugar (from sugar beet)	Jansen (2005); LCA Food Database (2003)	GWP
Maize starch, soy flour, sodium carbonate	Nemecek *et al* (2004); Althaus *et al* (2004)	GWP
Wheat, Wheat flour, Graham flour	LCA Food Database (2003a); Rosing and Nielsen (2003)	GWP
Free range eggs	ASDA Sustainability Manager (2007); Williams *et al* (2006)	GWP
Sugar (from sugar cane)	Ramjeawon (2004); Unilever Environmental Sustainability Manager (2007)	GWP, energy
Cocoa products	Dutilh and Chehab (1998); Afrane and Ntiamoah (2007)	GWP
Vegetable oils and soy lecithin	Shonfield (2005)	GWP
Berries, bananas, apples,	Wallen *et al* (2004)	GWP
Processes	**Source**	**Data**
Milk and yogurt processing	Cederberg (2003); UNEP (2000); Nielsen (2003); Feitz *et al* (2005)	Energy
Manufacturing energy	Unilever Hellendoorn factory Environmental Performance Report (2007)	Energy
Transport (lorry, ship)	DEFRA (2007)	CO_2
Refrigeration during transport	Unilever Lead Engineer Refrigeration (2008); McKinnon and Campbell (1998); International Institute of Refrigeration (2003); Magnum-lease (2008)	Energy
Refrigeration systems energies and refrigerant leakage data	Unilever Lead Engineer Refrigeration (2008); Unilever Ice Cream Refrigeration Expert (2007); IPCC (2005)	Energy, leakage
Manufacturing of glucose syrup from starch	Unilever Supply Chain Technologist (2008)	GWP
Wheat milling	LCA Food Database (2003b)	Energy
Supply Chain	**Source**	**Data**
Distribution centre energy	Burrows and Bassett (2006)	GWP
Distances and transported volumes information	Unilever product demand planners in each European market country (2008)	Distances
Cabinets and retail freezers	Unilever Lead Engineer Refrigeration (2008); Unilever Ice Cream Refrigeration Expert (2007)	Energy
Home freezers	Tribaluk (2008)	Energy
Transport home	Pretty *et al* (2005)	Distances, means of transport

Data related to the background system were mostly sourced from published literature. Efforts were made to select data from studies where geographical and technological conditions where similar to those in the system under study (especially relevant for ingredients). Other sources of information included internal LCAs (in the case of some ingredients) and expert opinion and calculations (e.g. for in-transit refrigeration). For ingredients, where more than one set of relevant data were available in the literature; these were used to calculate high and low scenarios, in an attempt to recognise the potential range of behaviours and practices which could occur at this stage in the chain. Data variability is also relevant at other stages of the chain (particularly retail and home refrigeration – Section 3). However, greater levels of variability were anticipated for ingredients recognising the natural fluctuations in energy and material inputs and yield which occur in bio-based systems; as such, attempts to quantify data variability were focussed here (Fig. 2).

3. Results

The results of the mass-balance approach are shown in Fig. 2; here the relative contribution of each life cycle stage to the total CF is illustrated. Ingredients contribute 33% of the overall impact of the study (31% in the low and 35% in the high scenarios), roughly half of which originates from dairy ingredients. Whilst the range of outcomes for ingredients appears rather limited, it should be noted that this would have been greater if B&J's specific data for dairy had not been available, and of course if potential land use and land use change impacts had been considered.

B&J's owned operations (ice cream manufacture) were found to be least significant, contributing 2% to the overall CF; mainly due to the fact that management steps have already been taken in the factory (including the purchase of green electricity).

With 46% of the total impact, refrigeration at the retail outlets (including refrigerant leakage) is by far the greatest contributor to the CF. Preliminary investigation suggests that freezers used in bigger retail outlets have a greater impact per litre of capacity than those used in small shops (predominantly due to energy use and refrigerant leakage rates). However, the result obtained for retail is more sensitive to the rate of throughput (i.e. the time spent in retail freezers) than the type of cabinet; current refrigeration technologies are similar across most European countries (IPCC, 2005). The estimated length of stay (Tab. 1) was based on the frequency of retail orders in the UK, assuming that this is representative of other European countries (given limited access to data for these other countries). However, if order frequencies vary greatly across Europe (e.g. between high and low volume markets), then the implications for the result are significant. Further work to refine the estimate for this life cycle stage is desirable; particularly to describe the range of outcomes for retail in different market countries and seasons.

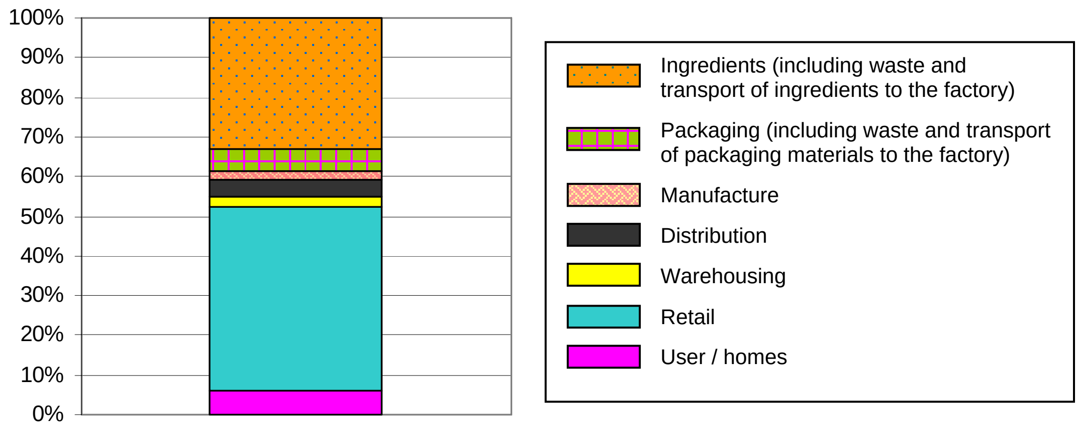

Fig. 2: Relative contribution of life-cycle stages for total annualised production of B&J's ice cream in Europe – Average scenario.

The remaining GHG contributions come from packaging (6%) (including secondary and tertiary packaging but not including waste disposal impacts (Section 2.2)), distribution (4%) (CO_2 from fuel combustion and GWP of refrigerant leakage), warehousing (3%) and consumer use (6%) (including transport to homes and refrigeration; however, this is only relevant for 'take home' products such as the 500ml tubs which are not purchased for immediate consumption).

4. Discussion

4.1 Mass-Balance assessment approach

The mass-balance approach employed in this work (and summarised in Tab. 1) offers a number of advantages for Carbon Footprinting in a business context. Specifically the ability to:

- understand impacts related to the totality of business operations which bring a portfolio of products to market, independent of functional unit;
- take management decisions even though there are methodological and data limitations which constrain our ability to calculate precise CFs;
- focus the majority of available resources on the management of impacts rather than measurement;
- make brand-based (total product portfolio) claims.

In standard (process-based) LCA methodology, a functional unit is defined so that all impacts are described relative to the product function (Baumann and Tillman, 2004). This is appropriate if the results are intended to highlight hotspots with a view to managing impacts related to a specific function. However, the approach is less relevant for businesses seeking to understand and manage impacts related to the totality of operations which bring a portfolio of products to market, *independent* of functional unit, and it is this total product portfolio which usually provides the context for strategic decision-making in business. Development of the mass-balance approach helps to address this by defining a reference unit which accounts for annualised production volumes of all ice cream flavours and formats (Section 2.1) thus eliminating the need for numerous CF studies, or the over-simplistic extrapolation of results which assumes that impacts associated with one product are representative of the whole product portfolio. On the other hand, if the need to assess an individual product (i.e. a flavour in a specific format) arises at any point, the mass-balance study will prove inadequate since there is no viable way to disaggregate the data to the product level. The same restriction applies if the aim is to compare the impact of two or more products (e.g. flavours / formats etc.).

When applying standard LCA approaches in the agri-food sector, it is common practice for a single figure for GWP to be calculated (this may go some way to explain why the development of certain communication approaches, such as carbon labelling, have also focused on single numbers, e.g. The Carbon Trust carbon label currently operating in the UK). In fact, this entirely fails to acknowledge the range of likely outcomes which should always be expected, and which result from variability in technological, temporal, spatial and behavioural characteristics along any given value chain. In this study, high and low scenarios were considered for the ingredients stage (Tab. 1) in an attempt to recognise and begin to quantify this variability. The average of these two scenarios provides the general understanding (e.g. hotspots – Fig. 2) needed for pursuing management activities (though in this case, the conclusions of the contribution analysis are stable even considering the high and low scenarios), providing a single figure relevant for decision-making (e.g. offsetting). The range between the high and low scenarios for ingredients is openly acknowledged in communication with informed stakeholders (e.g. the NGO contract partner HIER[9] with whom B&J's has worked to come to a 'climate neutral' status following their three step approach) in the brand's attempts to share and improve its understanding and demonstrate transparency and commitment to reducing impacts.

In reference to the third point noted above, the mass balance approach illustrates a cost and resource efficient approach to studying an entire product portfolio (only one CF need be performed as opposed to >40 in each market country in the case of B&J's). However, it is also relevant to consider how

[9] HIER is a Dutch consortium of 38 NGOs, including the Dutch division of WWF, Greenpeace and the Red Cross, with a shared focus on climate impact reduction. HIER (2008) has developed a three step approach for businesses to produce climate neutral products and services.

businesses currently record the information which is required for carbon footprinting. This is especially relevant for the foreground life cycle stages, and has significant implications in terms of the resource needed for data collection. Most businesses do not currently measure and record GHG data at the level of individual products / SKUs. In fact, sub-metering of utilities in factories / warehouses etc is rare. Data requirements for the mass-balance approach tend to be more aligned to current business accounting systems. For example, site level energy data are more readily available than data related to individual product lines or factory machines. Purchased ingredients volumes and logistics information also tend to be aggregated for total annual production. The resource required for data collection when applying the mass-balance approach is therefore lower, with the important added benefit of greater data accuracy (since problems of allocating material and energy inputs recorded at the site level to individual flavours and formats of ice cream are minimised).

Having said this, availability of specific data along the life cycle remains problematic: data management systems in businesses are geared to management of business-to-business transactions (including ingredient and product quality specifications) and the logistics (increasingly just-in-time) of bringing finished goods to market. For this reason, data relevant for estimating environmental impacts such as GHG emissions are often stored in disparate systems or are entirely missing (especially related to activities occurring up and down stream from a company's owned activities). Whilst these data gaps can sometimes be filled with secondary data, as illustrated in this study, considerable effort is required to establish the quality and relevance of these data for the particular study in question. Significantly, data are almost always missing for minor ingredients (e.g. vanilla extract), though it's possible that the manufacture of such ingredients could be energy and therefore GHG intensive.

4.2 Management

B&J's has as a mission "to make great ice cream in the nicest possible way" (Ben & Jerry's, 2008a). Part of its strategy to achieve this mission involves addressing climate change (Section 1). The results of this study clearly indicate life cycle hotspots (ingredients and retail) enabling Ben and Jerry's to prioritise its management efforts and focus resources on initiatives in these areas; examples of current and planned activities include (Ben & Jerry's, 2008b):

- Dairy Farms: reduce the amount of fertilizer used for the production of feed; invest in energy efficient heat-cold exchangers to simultaneously cool milk and heat water; install wind turbines on some farms; and research alternative feed options that may help to reduce methane emissions from dairy cows.
- Retail: research and implement new refrigeration technologies for cabinets which will reduce energy consumption and/or utilise refrigerants which do not contribute to climate change.

Since neither of the life cycle stages identified as GHG hotspots through this work are under the direct control of B&J's, the examples given above are dependant on collaboration and partnership in the supply chain. Whilst less significant in terms of the contribution analysis, the stage which is directly owned by the brand should be easier to manage and examples of ongoing and planned activities in this area include:

- Factory: implement energy efficiency measures; purchase green electricity; develop on-site renewable power generation (solar, wind turbines and biogas digester to turn waste into energy).

Finally, after pursuing a range of measures to reduce its carbon footprint, such as those listed above, B&J's have decided to 'offset' the GHG emissions which remain using a Gold Standard Verified Emissions Reductions scheme, allowing the brand to claim 'climate neutrality'.

4.3 Communication

The concept of 'climate neutral' ice cream allows the brand to communicate its efforts to understand and manage the GHG emissions associated with ice cream in a simple and engaging way to consumers and other 'non-expert' stakeholders. The style of communication employed by B&J's is uniquely light- hearted. An 'on-pack' climate neutral message has been designed in an effort to connect with consumers on this issue, whilst avoiding extensive explanation. The message directs consumers to B&J's Climate Neutral web section which is intended as the main vehicle for communication on this topic since on-pack communications tend to be difficult, relying on consumers to read packets where space is often limited and a number of messages compete for attention. The website displays the results from this study (in pie-chart format) along with on-line games which aim to explain the GHG hotspots; e.g. 'Belchin' Bovines' explains the link between methane gas emitted by cows and global warming (Ben & Jerry's, 2008b). Information packs distributed to the media when the climate neutral initiative was launched contained "whoopee cushions" sporting phrases such as "Nice Dairy Air" and "Less wind, more wind farms".

Approaches to communicating CF results are sensitive to a series of pitfalls that could undermine the perceived credibility of the results as well as the brand's commitment to tackling climate change. For example, concepts such as "carbon equivalents" are not readily understood by consumers: whilst public awareness of terms such as "global warming" and "climate change" is high (particularly in countries such as the UK), deeper understanding of the science is far more limited (e.g. knowledge of the main greenhouse gases) (Anable *et al.*, 2006). Thus, the term 'climate neutral' makes clear reference to terms where consumer / public recognition is likely to be highest (e.g. climate change).

There is also the question of how to communicate variability and uncertainty of results. Whilst presentation of a range of values would be a more reasonable reflection of real life (Sections 2 & 3), communication of such ranges could result in confusion, risking the perceived credibility of the results, and potentially turning consumers away from the whole issue of climate change. Whilst it is tempting to communicate a single figure instead (as in current carbon labelling approaches) this could be misleading and also requires consumer understanding of terms such as 'carbon equivalents'. B&J's have so far avoided communication of variability and uncertainty with most audiences, though the variability of outcomes is accounted for in the brand's strategy and reduction activities through use of the *average* GHG figure as a baseline against which to benchmark reductions. So far, communication of variability has been limited to certain stakeholder groups (as described in Section 4.1). In these instances, the results of the annualised assessment can be scaled down to describe the impacts associated with a pint of ice cream since most people are more likely to identify with a single tub of ice cream rather than total annual production volumes. Since the mass-balance considers an amalgamation of various different products (i.e. flavours) that lose their identity once grouped, results normalised to a pint of ice cream are generic, describing a theoretical 'meta-product' composed of a weighted average of the ingredients used in all ice cream flavours in 2006. The meta-product represents the average of all current products but is not specific to any individual flavour or SKU.

5. Conclusion

The 'mass-balance' approach described in this paper offers numerous benefits for strategic management of impacts within a business context when compared to the standard process-based LCA. B&J's have used the results of the study outlined here to support on-going initiatives and to activate newly identified opportunities for GHG reduction. The Climate Neutral initiative offers some specific benefits in terms of reputation management and consumer and stakeholder engagement, allowing the brand to communicate simple, strong and consumer-focused messages. However, for brands such as Ben and Jerry's, it is also important to consider other factors, not just GHG emissions and recognise that conflicts may arise between these factors.

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Life cycle assessment of feeding livestock with European grain legumes

D. U. Baumgartner[1], L. de Baan[1], T. Nemecek[1], F. Pressenda[2] and K. Crépon[3]
[1]Agroscope Reckenholz-Tänikon Research Station ART, CH-8046 Zurich, Switzerland,
daniel.baumgartner@art.admin.ch
[2]CEREOPA, AgroParisTech, F-75231 Paris, France;
[3]UNIP, F-75008 Paris, France

Keywords: life cycle assessment, LCA, grain legumes, soya bean meal, feedstuff, livestock production, Europe

Abstract

European livestock production is highly dependent on soya bean imports from overseas. Besides environmental issues, i.e. long transport distances and the deforestation of the rainforest, soya bean cultivation in South America has lately been criticised for its negative social impacts, such as food shortening due to biofuel production and expulsion of small holders due to the increase of cultivated area. Using European grown grain legumes (pulses) for fattening animals seems a viable alternative, especially since only 2% of Europe's arable land is cultivated with them. But what are the environmental impacts of substituting soya with European pulses?

We assessed the environmental impact of replacing soya bean meal with grain legumes produced in Europe in the feed for pigs, broilers, laying hens and dairy cows in different European regions using the life cycle assessment (LCA) methodology.

There were no overall advantages from the feed alternative containing European grain legumes: While energy demand and the global warming potential were reduced by 1% to 9%, eutrophication potential was similar with the exception of pig production in Catalonia, where high NO_3-losses in connection with the cultivation of peas led to a higher impact. For ecotoxicity there was a tendency towards negative environmental impacts of the European grain legumes feed alternative.

Conclusively, the use of grain legumes produced in Europe decreased the environmental impact from transport and from land transformation compared with imported soya beans. However, the results are more determined by the whole composition of the feed formulas than by the replacement of soya bean meal by grain legumes. This should be considered in formulating the feedstuffs. Measures have primarily to be taken to reduce the environmental burden of the feedstuff production, but also optimising animal husbandry and manure management should be aimed for.

Introduction

An important part of the human diet in Europe consists of products of animal origin. At the same time, animal production is economically the largest branch of European agriculture. In 2002, 37 million tonnes of meat, 33 million tonnes of milk and 5 million tonnes of eggs were consumed in the EU-15 (EUROSTAT, 2007). Rearing the large numbers of animals needed to supply these products puts pressure on the environment by using non-renewable resources and by emitting nutrients and pollutants to water, soil, and air. Feedstuff production is one of the major processes contributing to these environmental impacts (Basset-Mens & van der Werf, 2005).

Today more than 70% of the protein sources for animal feed for the European Union are imported, mostly as soya bean meal from North and South America. The adverse environmental impacts of long transport distances, the conversion of rainforests into arable land and the cropping of genetically modified cultivars act negatively on consumers' acceptance. Cultivation of more grain legumes in Europe is thus expected to be an interesting alternative to the importation of soya bean meal, particularly since grain legumes, being capable of symbiotic nitrogen fixation, do not need any nitrogen fertilisation.

Previous studies on environmental impacts of pig production focussed on different production systems such as good agricultural practise, label production and organic agriculture (Basset-Mens & van der Werf, 2005) or different feeding scenarios, i.e. extrapolation of the present trend in soya bean meal use, formulation with domestic feed with low crude protein level and added synthetic amino acids and feed from organic production (Eriksson *et al.*, 2005). There are comparative LCA studies on milk production examining the differences between conventional and organic milk production (Cederberg, 1998; de Boer, 2003). However the use of grain legumes as feed is not a major consideration in these studies. Only a few LCA studies have been performed on chicken production. Katajajuuri (2007) assessed the entire broiler chicken chain up to a marinated and sliced broiler fillet at the retail shop. Ostermeyer *et al.* (2002) compared the environmental impacts of two diets with heighted methionine levels, either through the use of synthetic methionine or by increasing the soya bean meal content.

The aim of this study was to assess the environmental potential of the replacement of soya bean meal from overseas by European grain legumes in animal feed for different animal products in different European regions. Production systems, transport distances, and feed composition are some of the important differences of the chosen scenarios.

Method

Five case studies in four European regions were conducted to analyse the environmental impacts of introducing grain legumes into animal feed: pork production in North-Rhine Westphalia (NRW, Germany) and in Catalonia (CAT, Spain), chicken and egg production in Brittany (BRI, France), and milk production in Devon and Cornwall (DAC, United Kingdom) (Baumgartner *et al.*, 2008). The selection of these regions was based on their national importance in producing the respective animal products (Crépon *et al.*, 2005). For all five case studies, a life cycle assessment (LCA) was calculated, comparing different feeding alternatives. In the life cycle approach, all stages of the agricultural production were included: the production of inputs and infrastructure (e.g. production of energy, fertilisers, seeds, machinery, buildings), crop production (e.g. fertiliser and pesticide application, harvesting, crop processing and storage, land transformation), and animal production (e.g. transport of feeds, direct animal emissions, manure management). Finally, the environmental impacts (emissions and resource use) for producing one kg of meat, eggs, or milk were assessed. Slaughtering and processing of the animal products were not considered. The LCA calculations were performed with the Swiss Agricultural Life Cycle Assessment methodology (SALCA) as described in Nemecek *et al.* (2008).

In order to formulate the different feeding alternatives, an economic optimisation model (Pressenda *et al.*, 2006) was used. The obtained formulas provided the necessary nutrients for every animal category with a realistic feedstuff composition. The formulas contained five categories of feedstuffs: i) soya bean meal (origin: Brazil, USA, Argentina), ii) different protein rich feeds (e.g. rapeseed, sunflower and palm kernel meal, maize gluten feed; origin Europe, Asia, America), iii) peas and faba beans (origin Europe), iv) energy rich feeds (e.g. wheat, wheat middlings, barley, grain maize, beet and citrus pulp, cassava, oils; origin Europe, America, Asia), and v) mineral feeds (e.g. limestone, di-calcium phosphate, synthetic amino acids, vitamins; origin Europe). Dairy cows also had roughage feed (fresh or conserved grass) in their ration.

The following two feeding alternatives were compared in all case studies: i) SOY, standard feed formulas with soya bean meal (and in the milk case study with other protein rich feeds) as the major source of protein; ii) GLEU, alternative feed formulas, where most of the soya bean meal was replaced by grain legumes from Europe (i.e. peas and faba beans) and different protein feeds. As grain legumes provide both protein and energy, a partial replacement of energy rich feeds also took place in those feed formulas. In the broiler chicken case studies two additional feeding alternatives were analysed: the SAA alternative, i.e. feed formulas containing higher levels of synthetic amino acids in combination with maize gluten meal and grain maize, but with almost no soya bean meal; and the short-SOY alternative, a more common chicken production system with a shorter fattening length (41 days instead of 60 days), where inclusion of peas instead of soya bean meal is not possible for nutritional reasons (Baumgartner *et al.*, 2008).

Results

Energy demand for producing eggs in Brittany

The main process steps determining the demand for non-renewable energy were lay hen housing, transport and energy rich feeds (Fig. 1). Feedstuff production accounted for about 45% of the total energy demand. The GLEU alternative had, compared with SOY, a favourable impact on the demand for non-renewable energy, with a 5% reduction. The main reasons were the reduced demand for energy for transport (- 28%) and production of energy rich feeds (- 23%). As for the other case studies, the reduced transport was due to the substitution of soya bean meal from overseas by European peas. For energy rich feeds, the reduced demand for energy stemmed from the altered composition of this category of feeds: In the GLEU alternative there was considerably less grain maize than in the SOY alternative. Grain maize has a comparatively high energy demand because of the grain drying after harvest. However, this positive effect is decreased due to the feedstuff substitutions in the category of protein rich feeds, where an increased use of sunflower meal and maize gluten is accompanying the introduction of peas into the GLEU formula. Both have a higher energy demand than the soya bean meal they are substituting, reducing the advantages of the replacement of soya beans.

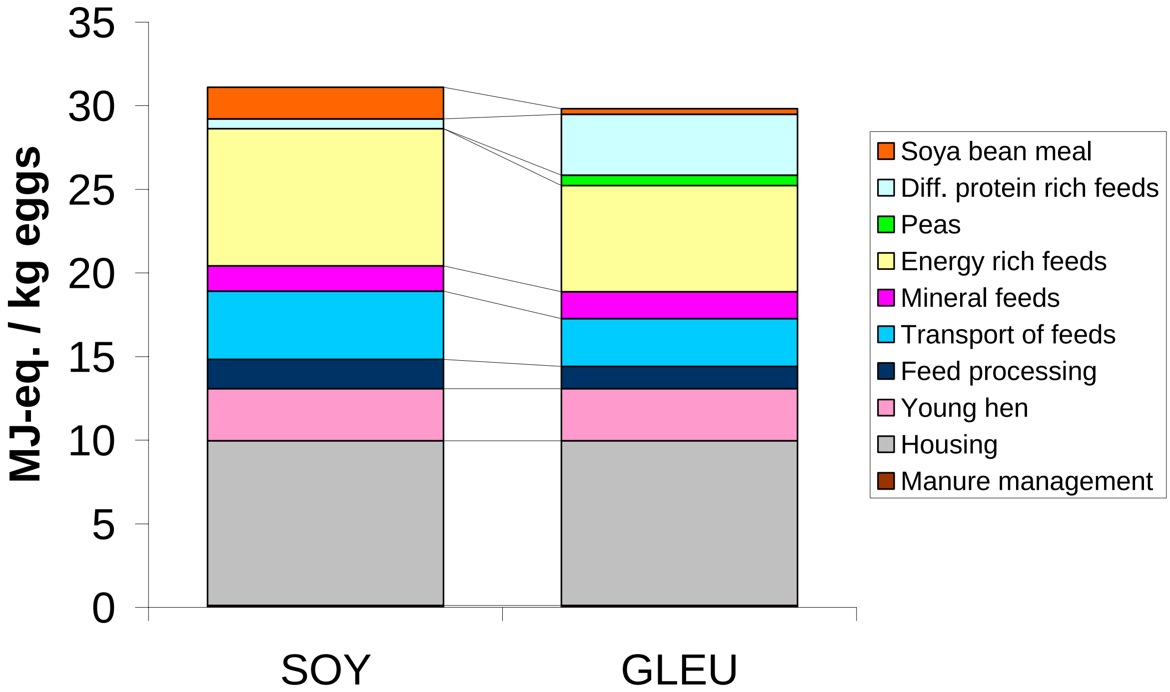

Fig. 1: Demand for non-renewable energy resources for producing one kg of eggs in Brittany (FR) with the two feeding alternatives, soya bean meal from overseas (SOY) or European grain legumes (GLEU).

Global warming potential for producing chicken meat in Brittany

Feedstuff production accounted for 70% of the global warming potential (GWP) of chicken meat production. The main difference between the feeding alternatives was the CO_2-release from land transformation, mainly for soya bean production in Brazil (Fig. 2). The GLEU alternative, containing very little soya bean meal and oil, had the lowest GWP of all four alternatives, whereas the short-SOY alternative, with the highest amount of soya, showed the highest GWP. This is also reflected in the impacts of transport, where short-SOY has the highest GWP. However, this alternative had, compared to all other alternatives, a decreased GWP for the process housing due to the higher productivity of the system. The GLEU alternative was favourable compared to SOY, through less transport of feeds and the absence of grain maize in those formulas, which have a high GWP due to the drying process. The increase of rapeseed meal and sunflower meal in protein rich feeds as well as peas led to a higher GWP for these process steps, decreasing the positive effects of less GWP from transport and energy rich feeds. Finally, the SAA alternative had a diminished GWP from transport of feeds, but a higher GWP from mineral feeds and the protein feeds replacing the soya bean meal. The reasons are an increase of the use of synthetic amino acids for the mineral feeds, the partial exchange of wheat by grain maize in the feedstuff group energy rich feeds and the introduction of maize gluten as a protein rich feed. Due to the grain drying after harvest grain maize has a comparatively high energy demand. Maize gluten is in its production much more energy intensive than soya bean meal.

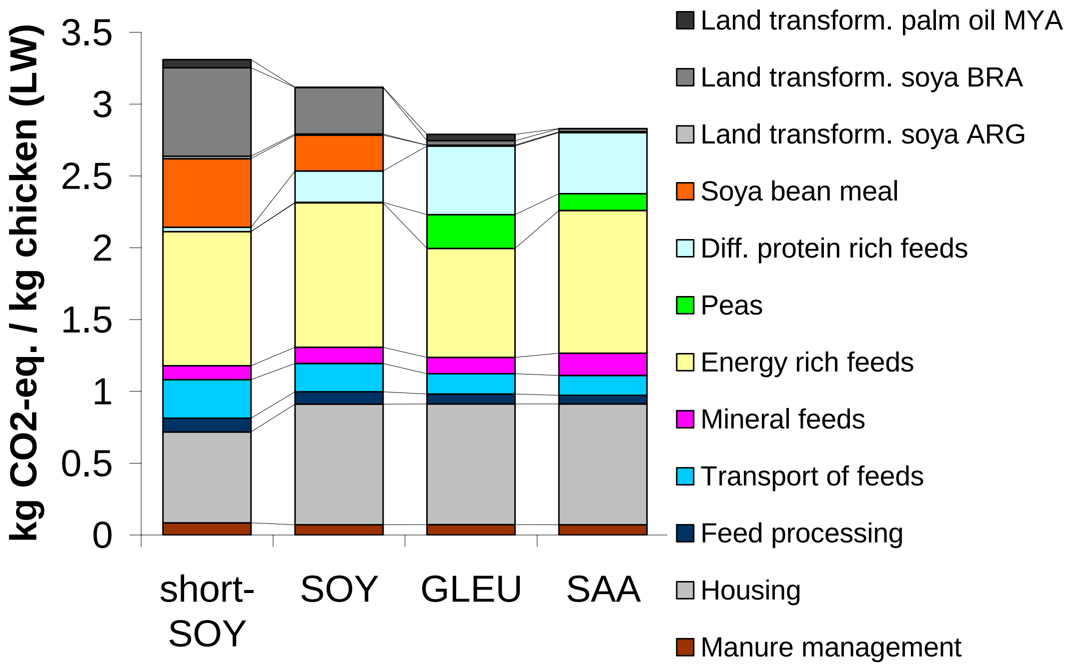

Fig. 2: Global warming potential (100a) for producing one kg of chicken (live weight: LW) in Brittany (BRI) with the four feeding strategies SOY (soya bean meal from overseas), GLEU (European grain legumes), SAA (synthetic amino acids), and short-SOY (short fattening length). MYA: Malaysia; BRA: Brazil; ARG: Argentina.

Eutrophication potential for producing pork in Catalonia

The incorporation of peas in the pig diet for Catalonia had, compared with the standard feeding (SOY), negative effects on the eutrophication potential. It was increased by 17% (Fig. 3). There was a slight reduction of the eutrophication potential for the energy rich feeds, but the main difference between the two alternatives lied in the increased eutrophication caused by nitrate losses in pea cultivation. In the LCA approach, all nutrient losses, from the harvest of the precedent crop to the harvest of the assessed crop (here spring peas), were attributed to pea cultivation. Thus, although peas were not fertilised, high nitrate leaching occurring prior to sowing and during mineralisation of organic matter after the cultivation, lead to an increased eutrophication of the GLEU alternative, especially due to the high incorporation rate of peas and the low yield levels of peas in Catalonia.

Compared to other regions in Europe, pig production in Catalonia showed a comparatively high eutrophication potential. This is due to a lower feed conversion rate, implicating an increased use of feed raw materials and increased losses of nutrients through excretion, and an unfavourable manure management (ammonia emissions from an uncovered slurry lagoon).

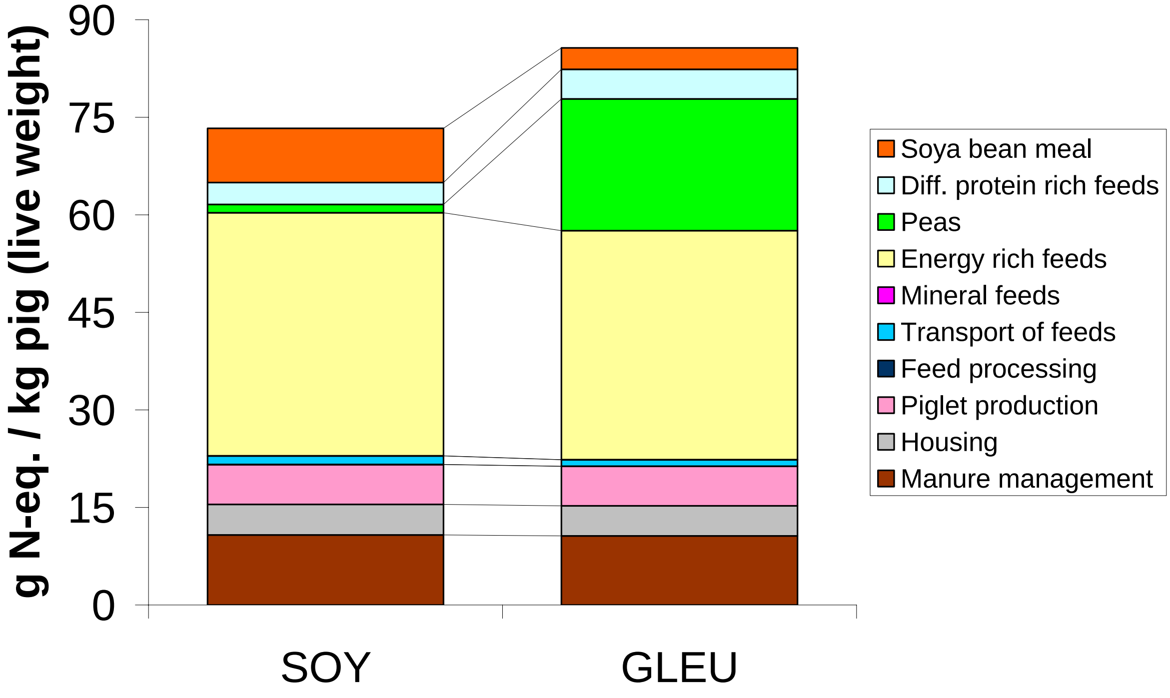

Fig. 3. Eutrophication potential per kg pork produced in Catalonia (CAT) with soya bean meal from overseas (SOY), European grain legumes (GLEU).

Overall results for all case studies

Overall, the environmental impacts of the GLEU alternative ranged from very favourable to very unfavourable compared with the SOY (Tab. 1). The results are classified in three environmental impact groups: resource use, nutrients and pollutants, as defined by Nemecek *et al.* (2005).

Feedstuffs contributed greatly to the environmental impact of animal products. In nearly all case studies, feedstuff production (crop production, transport, and processing) accounted for more than half of the energy demand and the eutrophication potential (nutrient enrichment), for about two-thirds of the global warming potential, and for most of the ecotoxicity. For dairy cows, the impact of concentrate feeds on the environmental burden was still high, but was slightly lower because the cows, fed mostly on grass and grass silage, consumed less concentrate feed than other animal categories.

Introducing grain legumes into animal feeds reduced the demand for non-renewable energy in all case studies except in North Rhine-Westphalia, where the GLEU alternative was similar to SOY (Tab. 1). The favourable effect of the GLEU alternative results from reduced transport and from the fact that pea and faba bean production is less energy intensive than the combination of soya bean meal and energy rich feeds that they are replacing. Global warming potential (GWP) was reduced in all case studies except for Catalonia. The transformation of Brazilian rainforest and Argentinean savannas into soya bean cultivation areas leads to large releases of CO_2 from biomass and soils. Replacing soya bean meal with grain legumes had little effect on the nutrient-driven impacts with exception of the eutrophication potential in pork production in Catalonia. There the low yield level of peas in combination with a high incorporation rate of them led to the negative impacts of the GLEU alternative (see above). Throughout all case studies the results for terrestrial and aquatic ecotoxicity ranged between a similar to unfavourable effect of GLEU compared with SOY (Tab. 1). Only in the milk case study the aquatic ecotoxicity of GLEU was slightly reduced. For the terrestrial ecotoxicity (according to EDIP97 methodology) cereals, rapeseed meal and peas dominated the results, while soya bean meal contributed little to this impact category. The reason lies in the applied active ingredients (pesticides) during the cultivation of the above mentioned crops. The detailed analysis showed that two active ingredients were responsible for the largest part of the terrestrial ecotoxicity according to

EDIP97, namely i) the fungicide propiconazole, which is used in cereals and ii) the insecticide lamda-cyhalothrin, which is applied in pea, oilseed rape and cereal cultivation. Since the results for ecotoxicity are very dependent on the applied active ingredients and the method chosen to assess them, a careful interpretation of the results is required.

Tab. 1: Environmental impact of feed formulas with European grain legumes (GLEU alternatives) as a percentage of feed formulas with soya bean meal from overseas (SOY) for all five case studies (per kg animal product) in North Rhine-Westphalia (NRW), Catalonia (CAT), Brittany (BRI) and Devon and Cornwall (DAC) (++ = very favourable, + = favourable, 0 = similar, – = unfavourable, – – = very unfavourable; EDIP and CML are two alternative ecotoxicity impact assessment methods.)

Region **GLEU in % SOY**	Reference flow	**NRW** **Pork** kg LW	**CAT** **Pork** kg LW	**BRI** **Chicken** kg LW	**BRI** **Egg** kg eggs	**DAC** **Milk** kg ECM
Resource use-driven impacts	Energy demand [MJ-equivalents]	0	+	+	+	+
	Global warming potential [kg CO_2-equivalents]	+	0	+	++	0
	Ozone formation [g Ethylene-equivalents]	0	–	0	+	0
Nutrient-driven impacts	Eutrophication [g N-equivalents]	0	–	0	0	0
	Acidification [g SO_2-equivalents]	0	0	0	0	0
Pollutant-driven impacts	Terrestrial ecotoxicity EDIP [points]	0	–	–	–	0
	Aquatic ecotoxicity EDIP [points]	0	–	0	–	+
	Terrestrial ecotoxicity CML [points]	– –	– –	0	–	0
	Aquatic ecotoxicity CML [points]	– –	0	0	–	0
	Human toxicity CML [points]	0	0	0	0	0

Discussion and Conclusions

Replacing soya bean meal with European grain legumes in feedstuffs was expected to improve the environmental performance of livestock production. The results of the five case studies on meat, egg, and milk production revealed that this replacement did not lead to an overall environmental improvement. Clear benefits could only be found regarding the resource use-driven impacts due to less transport, reduced incorporation of energy rich feeds and absence of land transformation. There was little effect on nutrient-driven impacts, as the positive effects of the reduced use of soya bean meal and energy rich feeds were often (over) compensated by the negative effects of the cultivation of the grain legumes themselves or the accompanying protein rich feeds, especially sunflower and rapeseed meal. For the pollutant-driven impacts, the introduction of grain legumes in feedstuffs tended to have negative impacts. Again, the reason lies in the crop production, where the feed ingredients replacing the soya bean meal involve using particularly harmful pesticides. However, these results should be checked with improved ecotoxicity assessment methods, as in some case studies they vary considerably between the methodologies applied.

It has to be stressed, that replacing soya bean meal by grain legumes changes the whole composition of the feed formulas and not only the part of the protein rich feeds. Consequently, the results are more determined by the whole composition of the feed formulas than by the replacement of soya bean meal by grain legumes.

Having diverging results throughout the different environmental aspects highlights the importance of a holistic approach to the evaluation of the integration of European grain legumes in animal feed. This enables to detect alterations from one environmental problem to another. As the feedstuff production has a major share in the environmental impact of animal products, improvements should target this part of the life cycle. As a possible measure we propose the integration of environmental criteria into feedstuff formulation models, allowing the optimisation of feed formulas in terms of economic and environmental aspects.

The following factors have been identified in helping to improve the environmental performance of livestock production:

- Domestic feedstuff production or import from neighbouring countries is favourable.
- Feedstuffs that need low levels of inputs for crop production and processing are favourable. Here, it is important to consider inputs in relation to yield levels; lower yields often lead to higher emissions per unit of the commodity.
- Energy rich feeds are used in large amounts in feed formulas (with exception of dairy cows). Consequently, improving the environmental performance of their cultivation lessens the environmental burden in animal production.
- Transformation of natural landscapes into cropland should be avoided to reduce the GWP and to maintain biodiversity, which was not considered here.
- Improved feed conversion of animals reduces the consumption of feedstuffs and hence the overall environmental impact of animal products.
- Higher productivity of the animal production system, i.e. higher amounts of product output in the same period lessens the environmental impact of animal products.
- Manure management can be improved (e.g. by covering the slurry lagoon, adjusting the timing of slurry spreading and use of appropriate spreading techniques).

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Comparing options for pig slurry management by Life Cycle Assessment

Lopez-Ridaura S. [1,2], Deltour L. [1,3], Paillat J.M. [1,4] , van der Werf H.M.G.[1]
[1] INRA, UMR Sol Agro&hydro systèmes Spatialisation, F35042 Rennes
[2] INRA, UMR Innovation, UMR951, F34060, Montpellier
[3] ENESAD Dijon
[4] CIRAD, UpR Recyclage et risque, F34398 Montpellier
ridaura@supagro.inra.fr

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Abstract

Excess manure from intensive livestock production is a recognised environmental hazard as its mismanagement threatens the quality of water resources and contributes to emissions of NH_3, CH_4 and N_2O. For these reasons, farmers search for options to reduce environmental impacts of excess manure, while remaining productive and maintain their economic viability.

In this study we compare several scenarios for excess pig slurry management using Life Cycle Assessment. Scenarios include the collective transfer of slurry versus its biological treatment (i.e. in either collective or individual stations), the covering of slurry storage tanks (i.e. uncovered, natural crust and PVC cap) and different methods of slurry application to crop land (i.e. injection, surface spreading by trailing hose with and without tillage and splash plate).

Transfer of slurry has lower eutrophication and acidification potential than the individual or collective treatment of slurry due to lower NH_3, it also has a better performance in terms of energy use as it treatment consumes large amounts of electricity in the treatment process while transfer slurry represents a net saving of energy due to the substitution of fertilisers. Covering slurry tanks can reduce eutrophication and acidification potential by up to 70%, due to the reduction of NH_3 emissions and reduces energy use by 8%, due to greater fertiliser substitution. Injection represents the best technique for slurry application to crop land as it reduces eutrophication and acidification potential by 32 to 74% relative to surface spreading due also to reduced NH_3 emission. Extra energy needed for the injection of slurry is offset by the increased substitution of fertilisers due to reduced NH_3 emission.

An optimal system for slurry management would include the transfer of slurry for its use in substitution of fertilisers, covering of slurry tanks with a PVC cap and the injection of slurry. However, the economic and organisational feasibility of this system should be evaluated. Also, a possible increase of N_2O emission due to slurry injection should be further investigated.

Introduction

In Europe, livestock commodities represent the highest value of agricultural production for most countries (FAO, 2006). The intensification of livestock production in the last decades has been accompanied by its dissociation from crop production, as it substantially relies on imported feed for its economic profitability. Such imports have generated new challenges related to the treatment and disposal of manure and slurry, as increased nutrient concentrations on crop fields and in ground and surface water threaten the ecological stability of regions where intensive livestock production takes place. Moreover, gaseous emissions (NH_3, N_2O and CH_4), resulting from animal buildings, manure and slurry storage and spreading on crop land, also represent an important environmental burden associated with intensive livestock production. CH_4 and N_2O are powerful greenhouse gases (Kroeze, 1994; Houghton *et al.*, 2001). NH_3 is responsible for acidification of rain and of the environment and for the formation of aerosols (ApSimon *et al.*, 1987; Fangmeier *et al.*, 1994), it also contributes indirectly to N_2O emission by soils (IPCC, 2006).

Brittany, in the West of France, concentrates 40% of the country's intensive livestock farming, producing 56% of the country's pigs, 31% of its poultry and 21% of its dairy products (Savelli and Cebron, 2006). Intensive livestock production has also positioned Brittany as one of the most polluted regions in France, especially with respect to nitrate in ground and surface water, as organic and mineral Nitrogen (N) applied largely exceeds crop needs. It has been calculated that as much as 103 000 tons of N (30% of the total N applied) are applied in excess of crop needs, 65% of which come from animal manure (Cebron and Ferron, 2003).

As part of the implementation of policies related to the Nitrate Directive of the European Union (91/676/CEE), districts (*cantons*) in France have been classified in relation to their vulnerability to water pollution by N used in agriculture. Structural Surplus Zones (*Zones d'Excedent Structurel*: ZES) have been defined as zones where the production of N in the form of animal manure surpasses a threshold of 170 kg ha^{-1} $year^{-1}$ over the spreadable area. In Brittany, 104 out of 187 cantons are classified as ZES. In the ZES, livestock farms exceeding a certain production of N as animal manure (between 12 500 and 20 000 kg per year, depending on the canton) must develop a plan for the disposal of the excess N, to reduce its environmental impact by either treating the slurry or transferring it outside the ZES for its application to crop land (MIRE, 2004).

In the Southeast of Brittany, a group of pig farmers in ZES, producing 41 tons of excess N in the form of slurry, have developed a collective transfer and spreading plan. In the transfer plan, almost 7 000 m^3 of slurry would be transported (over 40 km) and applied in substitution of mineral fertiliser on crop land, belonging to farmers in a region with less than 140 kg animal N per ha of spreadable area.

The objective of this study was to explore the impact of different technical options for excess slurry management and compare their environmental performance in order to conceive an optimal system of slurry management with the lowest environmental impact.

Methods

In this study we compare several options for excess pig slurry management using Life Cycle Assessment. First we described a base scenario and the calculation of its environmental performance in relation to four impact categories (i.e. Eutrophication, Acidification, Climate Change and Non-renewable Energy Use) and then, we describe different options for excess slurry management such as its treatment in either individual or collective slurry treatment plants as well changes in relation to the cover of storage tanks and the application of slurry to crop fields. The functional unit used to compare the scenarios is one cubic meter of slurry either treated or transferred.

The reference scenario

The reference scenario includes the on-farm storage of slurry, its transport to the spreading area, its intermediate storage, and its injection into crop land. On farm, slurry is stored in circular uncovered tanks of reinforced concrete. To calculate the average level of slurry in the storage tanks and the residence time of one m^3 of slurry (82.2 days), we have considered that the capacity of the tank corresponds to eight months of slurry production (as stipulated by law in the case of spreading), that the production of slurry is continuous throughout the year and that the outflow from the tank is dictated by the spreading calendar of crops. Main crops include cereals (wheat, maize, oats), receiving slurry between February and April; rapeseed, receiving slurry in September; and grassland, receiving slurry all year round. The distribution during the year is as follows: 25% of the annual slurry production is spread in February, 34% in March, 17% in April, 8% in June and 16% in September. Average distance between the pig farmers and the area receiving the slurry is 39.2 km and the transport is done with a 25 m^3 payload semi-trailer truck. Once in the spreading area, the slurry is temporarily stored in a flexible tank (200 m^3) of PVC coated polyester (WINBAG™) and then injected to crop land in substitution of chemical fertilisers.

Estimated emissions of NH_3, N_2O and CH_4 during storage of slurry are based on emission factors measured in Brittany with a floating chamber and determined by infrared detection (CH_4, N_2O) and gas chromatography (NH_3), reported in Loyon *et al.* (2005, 2007). Because N concentration in the slurry used in our study (i.e. from finishing pigs only) is higher than that in the slurry used by Loyon *et al*. (2005, 2007) (i.e. mixed slurry), emission factors for NH_3 and N_2O were proportionally corrected in relation to the ammoniacal and total N content of the slurry, respectively. The residence time and the contact surface area of one m^3 of slurry in the storage tanks was calculated in relation to the size of an average storage tank (equivalent to 8 months of slurry production) and the calendar of spreading. A mass and nutrient balance was computed to calculate total gaseous losses and to know the characteristics of the slurry at the moment of application. NH_3 emission during slurry application is considered to be 20% of the ammoniacal N (Morvan and Leterme, 2001), but reduced by 80% due to its injection (UNECE, 1999; Basset-Mens *et al*,. 2007), resulting in 4% of the slurry ammoniacal N emitted as NH_3. N_2O emissions are considered to be 2% of the total nitrogen in the slurry (IPCC 2006).

Non-renewable energy use in the reference scenario includes the energy used for the transport of slurry to the spreading area (2.49 L of diesel m^{-3}) and for its injection to crop land (0.8 L of diesel m^{-3}). Resource use and emissions of pollutants associated with the production of the concrete and plastics (PVC) needed for the storage slurry, as well as those associated with the machinery needed for its application were based on BUWAL 250 (BUWAL, 1996). Because of its very minor influence, energy used for the production of trucks for transport was not included in this study.

As slurry is used in substitution of chemical fertilisers for crop growth, resource use and emissions occurred during manufacturing, transport and application of fertilisers were deducted from the environmental impact of the slurry transfer. After losses during storage and application of slurry, one m^3 of slurry applied to crop land provides 5.21 kg N, 3.31 kg of P_2O_5 and 4.96 kg of K_2O but, based on studies carried out within the region of study and in relation to the slurry composition, the actual Mineral Fertiliser Equivalents (MFEs) for these nutrients are 65%, 95% and 100%, respectively (i.e. 35% of N and 5% of P are considered to be immobilised in the soil) (Morvan and Leterme, 2001; Morvan *et al.* 2005 ; Linères *et al.* 2005). Resource use and emissions associated with the production, transportation and application of the chemical fertilisers are based on Davis and Haglund (1999) and van der Werf (unpublished data), emissions associated with the application of these fertilisers are considered to be 2% for NH_3 (ECETOC, 1994) and 1% for N_2O (IPCC, 2006).

The impact assessment

In LCA, besides the direct emissions and resource use, energy use and emissions occurred during extraction of raw materials, their transport and processing (i.e. indirect emissions and resource use) are included in the comparison of scenarios. For the quantification of such indirect resource use and emissions, the BUWAL 250 (BUWAL, 1996), ETH-ESU (Frischknecht and Jungbluth, 2004) and IDEMAT (TUDelft, 2001) databases were used as implemented in SimaPro 6 (PRéConsultants, 2001).

Total (direct and indirect) emissions and resource use are aggregated and expressed in terms of three impact categories (Guinée *et al.*, 2002): eutrophication (in kg PO_4 –eq.), acidification (in kg SO_2 –eq.) and climate change (in kg CO_2 –eq.). Energy use is expressed in terms of non-renewable energy use (in MJ of Low Heating Value (LHV)-eq.).

These potential environmental impacts are calculated from resource use and emissions of individual substances, which are multiplied by a characterisation factor for each impact category to which they may potentially contribute (Heijungs *et al.*, 1992). Characterisation factors are substance-specific, quantitative representations of the additional environmental pressure per unit emission of a substance (Huijbregts *et al.*, 2000). The characterisation factors used in this study are given below for each impact category.

Eutrophication covers all potential impacts of high environmental levels of macronutrients, in particular N and P. As recommended by Guinée *et al*. (2002), eutrophication potential was calculated using the generic factors in kg PO_4-equivalents: NH_3: 0.35, NO_3: 0.1, NO_2: 0.13, NO_x: 0.13, PO_4: 1. Acidifying pollutants have a wide variety of impacts on soil, groundwater, surface waters, biological organisms, ecosystems and materials (buildings). As recommended by Guinée *et al.* (2002), acidification potential was calculated using the average European factors by Huijbregts (1999) in kg SO_2-equivalents, NH_3: 1.6, NO_2: 0.5, NO_x: 0.5, SO_2: 1.2. Climate change was defined here as the impact of emissions on the heat radiation absorption of the atmosphere. As recommended by Guinée *et al.* (2002), Global Warming Potential for a 100 year time horizon (GWP_{100}) was calculated according to the GWP_{100} factors by IPCC (IPCC, 1997) in kg CO_2-equivalents, CO_2: 1, N_2O: 310, CH_4: 21. Finally, non-renewable energy use refers to the depletion of energetic resources. Non-renewable energy use was calculated using the Lower Heating Values (LHV) proposed in SimaPro v. 6 for: crude oil (42.6 MJ kg^{-1}), gas from oil production (40.9 MJ m^{-3}), natural gas (35 MJ m^{-3}), uranium (451000 MJ kg^{-1}), coal (18 MJ kg^{-1}), and lignite (8 MJ kg^{-1}) (PRé Consultants, 2001).

The alternative scenarios

Treating slurry in individual or collective stations

Two scenarios involving treating excess slurry were considered including either collective or individual treatment stations. The slurry treatment is of the aerobic or biological type (nitrification/denitrification), with previous separation of the solid and liquid fractions of the slurry with a centrifuge and the re-circulation of sludge. The solid fraction is composted for 9 weeks, involving the addition of 3% of straw and mechanical turning. Compost is then transported to a cereal production region at 200 km distance for its utilisation in substitution of fertilisers. A detailed description of the treatment process is described in Lopez-Ridaura *et al.* (2007a, 2008). The average abatement efficiency of the treatment is of 70% of the total N and 90% of the ammoniacal N (Loyon *et al.*, 2005).

In the collective treatment scenario, slurry is transported to an average distance of 12.1 km while in the individual treatment, no transport is required. Also, for the collective treatment an average on-farm storage time is 46.2 days plus 25 days storage in the treatment station itself, for the individual treatment, only 25 days of storage are considered. NH_3, CH_4 and N_2O emissions during storage and treatment are based on Loyon *et al.* (2005, 2007) and corrected for N content of the slurry as for the reference scenario. Nitrogen losses during composting of the solid fraction, essentially in the form of NH_3, the quantity (123 kg of compost m^{-3} of slurry) and characteristics of the final product were based on Le Bris *et al.* (2005). 50% of the ammoniacal N in the compost is considered to be lost in the form of NH_3 (Basset-Mens, *et al.* 2007) while 1% of the total N is lost in the form of N_2O (IPCC, 2006).

Electricity used for the treatment (centrifuge, aeration, pumping) is 18.7 kWh m^{-3} (Levasseur *et al.*, 2003), and diesel used for the transport of slurry to the collective station is 0.76 l of diesel m^{-3}; the transport of compost from individual or collective slurry treatment stations to the application area consumes 1.51 l of diesel per 123 kg of compost produced by one m^3 of raw slurry and the spreading of 123 kg of compost consumes 0.079 l of diesel (van der Werf, unpublished data). MFEs of compost for N, P and K is considered to be 10%, 88% and 100%, respectively.

Covering Storage tanks

In the reference scenario, storage tanks were considered to be uncovered and, as reported in Lopez-Ridaura *et al.* (2007a, 2008), emissions of CH_4 and N_2O during storage were important contributors to Climate change, and NH_3 emission during storage was the most important contributor to acidification and eutrophication. Moreover, these emissions of NH_3 reduced the amount of N applied to crops per m^3 of slurry transferred and therefore decreased the possible substitution of chemical fertilisers. A possible option to reduce such emission during storage is the covering of storage tanks. Two options were evaluate in this study (i) allowing the formation of a crust on the slurry stored and (ii) the cover of storage tanks with a PVC cap.

If slurry has a high dry matter content and is not disturbed during storage, a natural crust can be formed (van Caenegem *et al.* 2005). A natural crust reduces the contact area of slurry with the atmosphere as well as the effect of wind on the emissions of NH_3. Based on Sommer *et al.* (1993), Hörning *et al.* (1999) and Xue *et al.* (1999), we have considered that the crust reduces the emissions of ammonia by 15%. We have not considered any reduction of CH_4 and N_2O emissions by the crust as the slurry will have to be stirred before its transfer and all the CH_4 and N_2O will be released to the atmosphere.

A conic PVC cap is also used to cover slurry storage tanks and it can help to reduce NH_3 emissions up to 80% as it reduces the exchange of ammonia between the slurry and the atmosphere by halting the effect of wind (Dux *et al*. 2005). However, as the PVC cap is not airtight we have considered that there was no effect on the emission of CH_4 and N_2O in relation to the reference scenario. To cover an average tank of 100 m^2, 265 m^2 of PVC are needed because of its conic form; considering a yearly storage of 7665 m^3 per year and a life span of 15 years for the PVC cap, 0.002 kg of PVC are needed per m^3 of slurry.

Application of slurry

In the reference scenario we consider that slurry will be injected in the soil however, the availability of machinery for injection is limited in our case study and it is possible that part of the slurry is applied to the soil surface. Three additional scenarios have been evaluated: surface application by splash plate, surface application by trailing hose without tillage and surface application by trailing hose followed by ploughing.

Based on UNECE (1999) we have considered that splash plate does not have any reduction in the emission of ammonia (i.e. 20% of the ammoniacal N of slurry applied is lost to the atmosphere in the form of NH_3 (Morvan and Leterme, 2001)), trailing hose without tillage reduces ammonia emissions by 30% while trailing hose followed by ploughing reduces the emission by 50%. We have not considered any effect of the slurry application techniques on N_2O emissions due to a lack of consensus in the literature and we have used an emission factor of 2% of the total nitrogen in the slurry as proposed by IPCC (2006).

For diesel consumption we have considered that application of slurry with a splash plate consumes 0.4 litres of diesel, a trailing hose 0.5 L and a trailing hose followed by ploughing 0.8 L as the injection of slurry (see reference scenario).

In relation to substitution of fertilisers, after losses during storage and application and the MFE for different nutrients, one m^3 of slurry substitutes 3.22, 3.15 and 3.05 kg of N in the form of fertiliser for trailing hose followed by ploughing, trailing hose without tillage and splash plate, respectively. The application technique does not affect the substitution of P and K in the form of fertiliser.

Results

Tab. 1 shows the direct resource use and emissions for the different processes of the reference system as well as the avoided direct resource use and emissions due to the substitution of fertilisers.

Tab. 1: Main direct resource use and emission for 1 m^3 of raw slurry transferred for the reference scenario

	Material resources			Energy	Emissions			Avoided fertiliser		
	Concrete (kg)	Plastics (kg) (PVC ,PET)	Ag. Machinery (kg)	Diesel (litres)	CH_4 (kg)	NH_3 (kg)	N_2O (kg)	Amm Nit (kg N)	P_2O_5 (kg)	K_2O (kg)
Storage	7.7				5.44	0.652				
Transport				2.49						
Intermediate storage		0.005								
Injection			0.041	0.8		0.159	0.170			
Substitution of fertilisers			*-0.033*	*-0.24*		*-0.082*	*-0.053*	*-3.39*	*-3.14*	*-4.96*
TOTAL	**7.7**	**0.005**	**0.008**	**3.05**	**5.44**	**0.729**	**0.117**	**-3.39**	**-3.14**	**-4.96**

Tab. 2 shows the results of the impact analysis for the different categories for the reference scenario as well as the contribution of each process and of the different substances to each of the impact categories.

Tab. 2: Contribution of emitted substances and resources to four impact categories for the reference scenario, expressed per m^3 of slurry transferred

	Eutrophication Potential (g PO_4 –eq.)				Acidification Potential (g SO_2 –eq.)				
Substance	NH_3	NO_2	Other	**Total**	NH_3	NO_2	SO_2	Other	**Total**
Storage	228	0.4	0	228.4	1040	1	3	0	1044
Transport	0	17.6	0.2	17.8	0	67	14	0	81
Intermediate Storage	0	0	0	0	0	0	0	0	0
Injection	55.7	5.7	0	61.4	254	22	7	0	283
Fertilizer avoided	*-28.7*	*-15.7*	*-13.8*	*-58.2*	*-131*	*-60*	*-95*	*0*	*-286*
Total	255	8	-13.6	**249.4**	1163	30	-71	0	**1122**

	Climate Change Potential (kg CO_2 –eq.)				Non-Renewable Energy Use (MJ of LHV-eq.)				
Substance	CO_2	CH_4	N_2O	**Total**	Oil	Gas	Uranium	Other	**Total**
Storage	0.7	114	0	114.7	1.3	0.1	1.5	2.2	5.1
Transport	7.5	0.2	0.1	7.8	97	4	0.7	0.6	102.3
Intermediate Storage	0	0	0	0	0	0	0	0.3	0.3
Injection	2.7	0.1	52.8	55.6	33.1	2	2.2	1.7	39
Fertilizer avoided	*-18.2*	*-0.5*	*-32*	*-50.7*	*-75.5*	*-140.5*	*-10.4*	*-30.6*	*-257*
Total	-7.3	113.8	20.9	**127.4**	55.9	-134.4	-6	-25.8	**-110.3**

NH_3 is the most important contributor to eutrophication and acidification and it is specially emitted during storage of slurry. CH_4 is the most important contributor to climate change and also emitted mainly during storage of slurry. In terms of energy use, the transport of slurry to the application region is the most important process using energy, however the savings of energy corresponding to the avoided fertiliser compensate for the energy needed in transport and results in a net energy saving.

Fig. 1 shows the comparison of the individual and collective treatment in relation to the reference scenario. Treatment of slurry implies greater eutrophication and acidification than the transfer of slurry due to larger NH_3 emissions during storage and treatment, individual treatment performs better than collective treatment because storage time is considerably reduced. Individual treatment implies less climate change as it has shorter storage time and it does not involves any transport of slurry. In terms of energy use, the treatment of slurry consumes high levels of electricity and the compost by-product has a low substitution of fertilisers. Individual treatment consumes less energy than the collective treatment as no slurry transport takes place.

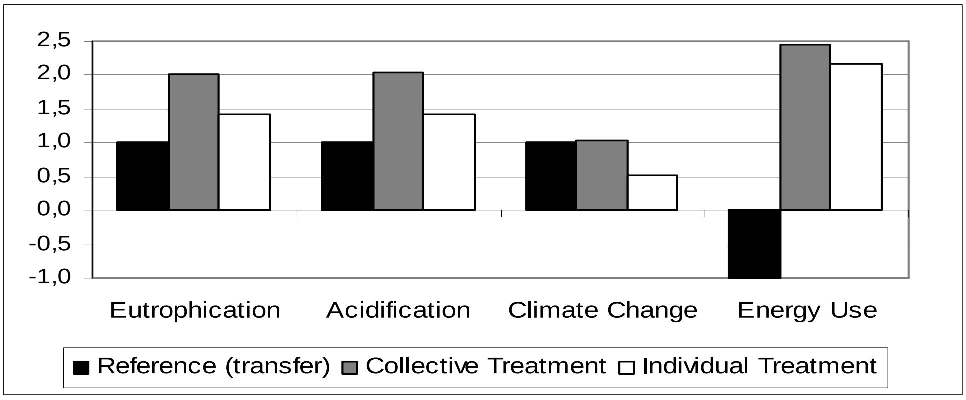

Fig. 1: Environmental impacts of three slurry management scenarios in relation to storage tank cover expressed as a fraction of impacts for the reference scenario

Fig. 2 shows the comparison of the scenarios including covering the storage tanks in relation to the reference scenario. Covering storage tanks strongly reduces Eutrophication and Acidification, especially when a PVC cap is used, as it reduces the emission of ammonia. Covering storage tanks also reduces energy use, as more N is available for the substitution of fertilisers per m^3 of slurry applied to crop land. This however causes a very slight increase in Climate change as we have considered that 2% of the N applied will be lost in the form of N_2O.

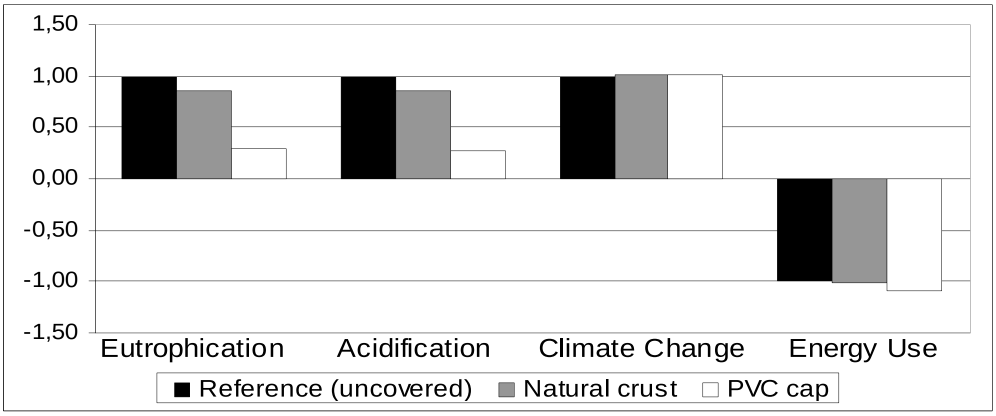

Fig. 2: Environmental impacts of three slurry management scenarios in relation to storage tank cover expressed as a fraction of impacts for the reference scenario

Fig. 3 shows the impacts for different slurry spreading techniques to crop land. Surface spreading without tillage increases Eutrophication and Acidification by more than 50% in relation to the reference scenario as more ammonia is emitted. Ploughing the after slurry application reduces Acidification and Eutrophication in comparison to surface spreading without tillage as ammonia emission is strongly reduced, however it remains more harmful to the environment than slurry injection. In relation to Climate change and Energy Use the four scenarios are nearly equivalent as, although substituting more chemical fertilisers in the case of injection and trailing hose with ploughing, the energy used for application increases when ploughing and injecting as well as the emission of N_2O.

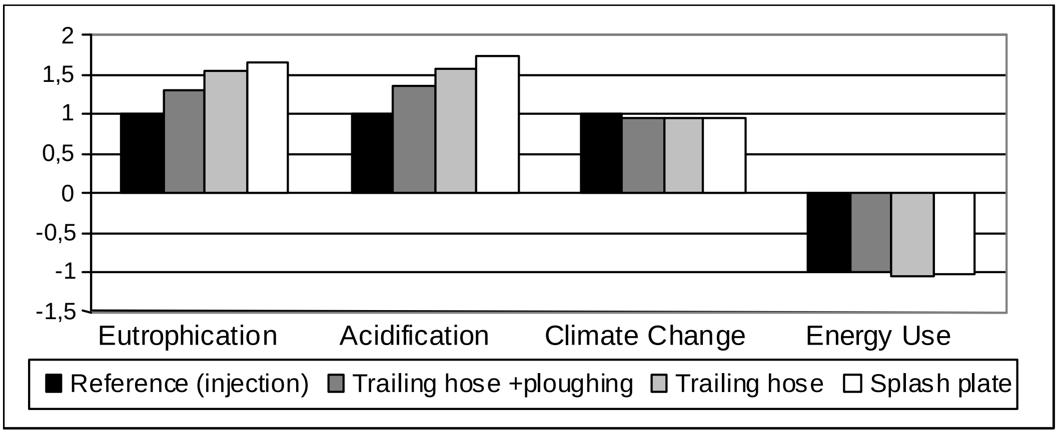

Fig. 3: Environmental impacts of four slurry management scenarios in relation to slurry application techniques expressed as a fraction of impacts for the reference scenario

Discussion and Conclusion

Manure transfer is a possible solution to reduce the environmental impact of intensive livestock production where nutrients in the form of manure exceed local crop needs. To identify possible advantages and burdens of manure transfer requires comprehensive studies considering its economic feasibility as well as its environmental performance.

We have compared scenarios for a pig slurry transfer plan with respect its treatment, the effect of covering of storage tanks and the effect of different modes of slurry application to crop land. The transfer of slurry for its utilisation in substitution of fertilisers has a better environmental performance than the treatment of slurry as it implies less eutrophication and acidification and a net saving in energy use due to the substitution of fertilisers. For climate change, individual treatment performs better as the storage time is reduced and therefore reduction on CH_4 emissions and no transport of slurry is considered.

Covering storage tanks strongly reduces the environmental impacts of slurry transfer, as less ammonia is emitted during storage (reducing acidification and eutrophication) and, since less N is lost, richer slurry is applied to crop land representing greater savings on chemical fertiliser and therefore reducing total energy use. These advantages are much larger for a PVC cap than for a natural crust as the former is much more efficient in reducing ammonia emission.

Regarding the application of slurry to crop land, injection is better than surface spreading as it strongly reduces ammonia emission while the extra energy costs of injection are compensated for by the greater substitution of chemical fertilisers. In relation to the environmental evaluation of the scenarios for slurry application, two aspects should be examined in further research: relative to surface application injection may increase N_2O emission.

With respect to N_2O emission we have used the IPCC (2006) emission factor for liquid manures (i.e. 2% of the total N) however, injection of slurry, relative to surface application, might increase N_2O emission as pores in the soil are filled with slurry and so-called anaerobic hot spots might be created increasing denitrification rates (Dendooven *et al.*, 1998). However, given the lack of sufficient published results quantifying this effect we did not consider it in this study.

Taking into account the scenarios tested in this study, an optimal system where slurry is transferred and applied to crop land in substitution of fertilisers, storage tanks are covered with a PVC cap and slurry is injected into crop land can be envisaged. However, the economic and technical feasibility of such a system in terms of extra investments (for the cover and the injector) should be evaluated, as well as its organisational feasibility. As part of our current research endeavours, we evaluate via a dynamic model the effect of different climatic conditions on the feasibility of the transfer plan and its environmental impact (Lopez-Ridaura *et al.*, 2007b), as rainfall, temperature and wind speed strongly affect the access to fields for slurry application and the gaseous emissions during storage and application of slurry.

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Environmental impacts and related options for improving the chicken meat supply chain

J.-M. Katajajuuri[1], J. Grönroos[2] and K. Usva[1]
[1]MTT Agrifood Research Finland, Jokioinen, Finland, [2]Finnish Environment Institute, Helsinki, Finland, juha-matti.katajajajuuri@mtt.fi

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Abstract
The environmental impacts of a typical Finnish broiler chicken fillet product were studied using a supply chain integrated life cycle assessment method. All essential production phases from parent stock and production of farming inputs to product distribution and sales in retail stores were included in the assessment. The results of the study clearly demonstrated the significance of the environmental releases caused by primary (incl. broiler chicken houses) production. For each impact category, most of the environmental impact along the chain originated from housing of broiler chickens and cultivation of feed ingredients. Broiler housing and feed production had the most impact on eutrophication and acidification due to nutrient run-off and leaching, and ammonia emissions from broiler chicken manure. To establish measures that could be taken to decrease environmental impacts of the supply chain, some scenarios are presented. The most deserving aspect, meriting further research, concerned the combination of ammonia and dust removing processes with heat recovery systems. Research in this area could result in several positive impacts in terms of decreasing ammonia emissions, improving broiler chicken health and saving energy.

1. Introduction
Increasing consumption and production underlie most of the environmental problems encountered in western countries. The European Commission adopted a Green Paper on Integrated Product Policy (IPP) in 2001, which seeks to minimise environmental impacts by looking at all phases of a products' life-cycle and taking action where it is most effective. In 2002, at the United Nations Johannesburg Conference, the issue of sustainable production and consumption was incorporated into the Plan of Implementation of the World Summit on Sustainable Development. In 2005 Finland launched its own national programme to promote sustainable consumption and production.

Food consumption represents around one-third of environmental impacts in Finland (Nissinen *et al*. 2007) as a result of the environmental impacts of agriculture. Moreover, because the Baltic Sea and inland waters are sensitive to nutrient releases, eutrophication is also an important impact category in Finland.

Consumption of broiler chicken meat is rapidly increasing in Finland. Most broiler chicken products in Finland are sold and eaten as honey-marinated fillets. In this study, life cycle assessment (LCA) was used to assess the environmental impacts of such fillets. Broiler chicken housing is centralised in Finland close to slaughterhouses, which has an effect on environmental impacts.

LCA results for broiler chicken products have not been published in scientific journals. Only comparative LCA was reported (Ellingsen & Aanondsen 2005), focusing on salmon farming. Furthermore, some LCA broiler chicken case studies were carried out in Sweden and the United Kingdom (Widheden *et al*. 2001, Williams *et al*. 2006).

Paying attention to the entire supply chain instead of individual production phases represents new possibilities for companies in the production chain. Product integrated sustainability and environmental management are essential elements for improving competitiveness of companies and their products and also reducing environmental impacts of their activities. In the Finnish Foodchain[10] LCA research programme, supply chain integrated LCA has been applied to the assessment of environmental impacts of foodstuffs of national importance.

The aim of this study was to increase knowledge on the environmental impacts of the broiler chicken production chain. Through a directed research approach the project aimed to identify potential measures to improve the environmental performance of Finnish broiler chicken production. In this study an effort was made to get the different parties involved in the supply chain to learn more about product-oriented environmental management and assessment of environmental impacts and related benefits, i.e. learning by doing. This provides a real possibility to seek continuous improvements in the supply chain.

2. Materials, methods and approach

2.1 Goal and scope definition of LCA

The aim of the study was both to increase knowledge on the environmental impacts of the broiler chicken meat production chain and gauge the contribution of the different production phases to energy consumption and environmental impacts in the system. The other aim of this approach was to recognise the potential measures to improve the environmental performance of Finnish broiler chicken production, particularly for the HK Ruokatalo Kariniemi brand supply chain. This was also a reason why the environmental impact assessment was based on actual production chain processes between 2003 and 2005. Most of the company's broiler chicken housing was situated in south-western Finland.

The functional unit (FU) of this supply chain integrated life cycle assessment was 1000 kg of honey-marinated sliced broiler chicken fillet produced and packed by HK Ruokatalo and purchased by consumers in retail shops.

Production chain processes comprised all essential production phases:

- rearing of young breeders and cockerels,
- production and transportation of the eggs,
- hatchery and transportation of the chicks,
- feed production for young breeders and cockerels, breeders and broilers,
- production of fertilisers and lime,
- production of peat litter,
- farming of broilers,
- transporting broilers to the processing plant,
- slaughtering and final meat product production,
- main consumer product package and other packaging materials,
- cultivation of turnip rape,
- production of turnip rape oil and marinade and
- product delivery in Finland and storage and selling in retail stores.

The assessment comprised primary energy consumption, direct and indirect emissions to the air, land use, amount of landfill waste as well as by-products and their applications. In addition, direct and indirect emissions to water were assessed. The environmental impacts were assessed in the form of climate change potential, aquatic eutrophication, acidification and photochemical ozone formation. Site-dependent characterisation factors were used for aquatic eutrophication and acidification (Seppälä et al 2006) assessment. IPCC 2000 factors were used to assess climate change and for tropospheric ozone formation the factors described by Hauschild *et al*. (2004) were used.

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[10] https://portal.mtt.fi/portal/page/portal/www_en/Projects/Foodchain.

The environmental impacts resulting from the broiler chicken production chain were also presented as total environmental impacts. In this approach, the same normalisation and weighting factors as used in the Finnish Eco-Benchmark method were used (Nissinen *et al*. 2007). The Eco-Benchmark takes into account five important environmental impacts (consumption of primary energy, global warming, acidification, eutrophication and tropospheric ozone formation), which were weighted according to their importance by a large group of Finnish environmental science experts.

Data for the system models were acquired from the field where possible. This focuses on the real operations and better quality and applicability of the results in, for example, formulating improvement options. Principles and benefits of the production chain-supply web-integrated LCA were more widely presented and discussed by Poikkimäki and Virtanen (2003).

Concerning allocations, it was attempted to obviate allocations by dividing processes into sub-processes. However, some allocations were carried out, and the appropriate principles were selected according to situation, the most important being presented in this chapter.

2.2 Life cycle inventory analysis

Data acquisition on cultivation

According to feed-use records for broiler farms, on average, one broiler chicken consumes 3.4 kg of feed during its life. Respectively, this corresponds to 2.5 kg feed per one dead weight kilogram of chicken broiler. Also during the preliminary phases of the production chain, i.e. in parent stock rearing, feed is consumed but in smaller amounts. Broiler feed comprises concentrated feed and farms' own feed (cereals). Concentrated feed includes different feed types according to the feeding phase. For both main feed types the key raw materials were cereals (wheat, barley and oat). In concentrated feeds soya also was included.

Cereals produced on the broiler chicken farm (home-grown cereals) and average Finnish crop farms (cereals for concentrated feed) were considered separately due to differences in farming practices, including use of fertilisers and soil soluble phosphorus concentrations. Data were acquired from the grain survey of the fodder company and from broiler farm records, except the data related to machinery, for which the main data sources were the work norms of the Work Efficiency Institute (Peltonen and Vanhala, 1992) and the Unit Emissions of Machinery Calculation System developed by VTT Technical Research Centre of Finland.

Production and emission data for soya production were obtained from Cederberg (1998), Cederberg and Darelius (2000), Kulay and Silva (2005) and Miller and Theis (2006).

Turnip rape production was considered in addition to cereals. The data source was ProAgria Agricultural Data Processing Centre ML Ltd. (unpublished database), which collects cultivation data directly from Finnish farmers. Turnip rape oil is the main component of the broiler chicken marinade.

Calculation models for emissions to air and water from cultivation

Atmospheric N_2O-emissions from the soil and emissions from agricultural lime were calculated using the IPCC emission factors (IPCC, 2000). The national factor for ammonia emission from mineral fertilisers (0.5%) was used (NH_3-N from applied mineral fertiliser-N; placement fertilisation with NPK fertilisers). Ammonia emissions from manure applied to soil were assessed based on the results from international studies.

A regression model based on field trials was used to assess nitrogen leaching from fields (Salo & Turtola 2006). In the model, nitrogen leaching is predicted by annual nitrogen balance, ΔN (formula 1) (Salo, 2005. Personal communication).

$$\text{N leaching (kg/ha/a)} = 5 + 0.16\Delta\text{N (kg/ha)} \quad (1)$$

Phosphorus leaching – including both dissolved reactive phosphorus (DRP) and particulate P (PP) - was calculated based on the method of Ekholm *et al*. (2005). Phosphorus leaching largely depends on total and soluble soil P concentration and on the degree of erosion, and is not as directly affected by fertilisation level as nitrogen. For broiler farms the phosphorus emissions were somewhat higher than for crop farms due to higher soil soluble phosphorus concentrations.

Data acquisition on fertiliser and lime production

Data on the consumption of primary energy and natural resources, and emissions to air and water in fertiliser production processes, was collected from the Finnish fertiliser producer. Most of the data were collected in 2002 but were updated in 2005. However, the reliability of the data was uneven, and a single representative fertiliser (nitrogen content of 20%) eco-profile was therefore formulated. Most of the environmental effects in the fertiliser production chain resulted from nitric acid production.

The consumption of primary energy and natural resources and emissions to air and water in the lime production process was based on information from the Finnish lime producer. Transport (including use of primary energy and emissions to air) and plastic packaging for fertilisers were included in the calculations.

Data acquisition on industrial feed production

Most of the feed used for broilers and breeders is processed industrially from cereals and soya. Data for electricity, heat, raw material mix and related production outputs, including air emission and side-stream volumes for broiler and parent stock feed production, were acquired directly from the industrial feed production site.

Industrial oil extraction and production of the soya meal were assessed based on similar production statistics from a soya meal producer. In the soya meal process soybean oil is produced and a respective allocation between oil and meal was calculated according to international stock prices of products in 2006.

Data acquisition on broiler rearing

The broiler chicken production chain consists of parent stock, hatching, rearing of broilers, slaughtering and meat processing. The data were collected from the actual operators in this chain.

Rearing of young breeders and cockerels, as well as egg production, takes place in separate broiler houses. Feed, water and litter consumption data and output data were obtained from the data records of HK Ruokatalo. Electricity and heat consumption during young breeder and cockerel production was estimated based on consumption data in broiler houses. Data concerning egg production were derived from surveying five egg producer farms.

For hatching, the actual production process was the main data source. Material flow data (eggs, chicks and waste material) and energy consumption data were based on data records of HK Ruokatalo. This information was validated and completed during the project using the company’s own follow-up data.

Broiler house process data in 2004 were acquired mainly from the records of HK Ruokatalo. This included data on feed, water and litter consumption in broiler houses and numbers of carcasses produced during the process. These data were secured through a questionnaire sent to a group of broiler producers. Data on the consumption of electricity and fuel in broiler houses were also obtained via this questionnaire. Information on manure handling practices was collected from 16 producers and the data were verified and validated together with the producers by phone and visiting the farms.

Heat consumption of broiler houses was on average 4.7 MJ/carcass kg according to the questionnaire data. However, the data for heat consumption in broiler houses varied and were validated using a theoretical model of heat consumption in a broiler house for 15 000 broiler chickens. The transport of young breeders and cockerels, eggs, chicks and broilers was included and modelled using real distances and types of transport used.

Estimations of ammonia emissions in broiler houses

Ammonia emission assessment was performed in two different ways: 1) using national and international studies (e.g. Arnold *et al*. 2006) on ammonia evaporation during broiler rearing and during manure storage, and 2) subtracting the amount of manure nitrogen in stored manure from the fresh manure nitrogen obtained from feeding nutrient balance calculations, where the data were obtained from farms (feeding, feed nitrogen content, number of broilers produced), from MTT Agrifood Research Finland (data on nitrogen content of birds) and from national manure analysis. Manure management data were received directly from farms and through expert interviews. According to the manure use records of broiler farms, 65% of manure is submitted to a manufacturer of organic fertilisers for further processing, and the rest is used as a fertiliser directly on broiler farms or on crop farms in the neighbourhood.

An ammonia emission factor of 0.15 kg NH_3/animal place/year for the rearing phase was used. For the whole manure management chain an emission factor of 0.18 was used. Both values were used in the NH_3-emission calculations.

Data acquisition on slaughtering and processing

Material flow data (raw materials, products and by-products) for slaughtering and processing plants were based on the follow-up data of HK Ruokatalo. These data were validated and complemented with measured data.

The amounts of energy and water consumption and waste water were measured for the whole plant. Energy consumption for honey-marinated sliced broiler chicken fillet was measured by splitting the entire process into component processes. The consumption of heat and electricity was measured wherever was possible. For some processes it was defined theoretically or, in some cases, estimated by experts at HK Ruokatalo.

Water consumption data were based on follow-up data of the plant. Washing water consumption in different processes was estimated using e.g. the records of cleaners' working methods. Heat production was based on the plant's own calculations.

The HK Ruokatalo broiler processing plant produces both boned and boneless products. Allocations between the products were done using meat mass in the products, not the total product mass. Using this principle, the different product types were treated equally. The transport of broilers to the slaughterhouse was included and modelled using real distances and actual transport means.

Data acquisition on marinade production

Data on use of electricity, heat, raw material mix and related production outputs, including air emission and side-stream amounts for industrial oil extraction and refining, were acquired directly from the industrial vegetable oil production site. In the turnip rape oil process, turnip rape meal is produced and the respective allocation between oil and meal was done according to international stock prices in 2006.

Data acquisition on packaging production

Package production data were acquired directly from the manufacturer. Material consumption, side stream materials and heat and electricity consumption data were based on data records of the manufacturer. Data from Plastics Europe (2006) were used for raw material production of packaging.

Data acquisition on product logistics and retail

Consumer products were assumed to be delivered throughout Finland according to current regional market shares. Emissions associated with product deliveries were modelled using realistic delivery routes with initial loading, retail stops, and final discharge of return load. Logistics were modelled in collaboration with a Finnish logistics company, and included retail product losses. The data for retail refrigeration were estimated using nominal electricity consumption of the refrigeration device and average product throughputs of the cold stores.

Data acquisition on energy production

Average Finnish grid electricity data from the year 2004 were used. The main sources of energy were fossil fuels 51%, nuclear 42%, wood 9% and hydro 6% (Statistics Finland, 2004). Local energy production, including steam and heat, was considered as it is used.

2.3 Improvement options

To establish the measures that could be taken to decrease environmental impacts of the production chain, some scenarios were defined together with the players in the chain. Environmental impacts of these scenarios were calculated.

The share of home-grain and industrial fodder

Of the total amount of broiler fodder, an average of 4% was home-grown, but its share is increasing compared with that of industrially processed feed. Based on foreseen changes in feed mix we calculated environmental impacts for five different feeding scenarios (Fig. 4). For the "common feed scenario" 100% of the fodder was industrially produced and for the four other scenarios the share of home-grown cereals varied (10%, 20%, 30% and 40%). In the last scenario (40%) home-grown cereals (wheat and oat) were stored in airproof silos without drying, instead of the commonly used method of drying and storing in open silos.

Alternative fuel in broiler houses

Most of the broiler houses were heated with light fuel oil in 2004, but also wood chips and pellets were used. The share of these renewable energy sources is increasing and for the "alternative fuel" scenario we investigated a broiler house heated with 50 % wood chips and 50 % wood pellets.

Heat recovery and alternative fuels in broiler houses

Broiler houses consume a lot of energy through ventilation and heating. Broilers also produce heat, especially at the end of their growing cycle. There is great potential to save energy using heat recovery systems. However, the dust content of outgoing air, treatment of condensed water and possible hygiene problems represent technical barriers to using such technology. This is why efficiency of heat recovery as low as 10% was used in this scenario.

3. Results

3.1 Environmental impacts of the current system

Broiler chicken housing and corresponding fodder production accounted for over 80% of all eutrophication impacts created by the entire production chain. This particularly concerned the crop cultivation needed to produce broiler chicken fodder, which contributed most to nutrient run-off and leaching (Fig. 1). However, these diffuse nutrient emissions are associated with high uncertainties. The contribution of the parent stock and related fodder production and hatching was 8-9% over all impact categories. The share of the marinade in the final product is much higher by mass than by corresponding environmental impact. The largest relative contribution made by the marinade (turnip rape oil) was in eutrophication, accounting for 4% of the total impact in that category.

Ammonia emissions from broiler chicken manure dominate the acidification impact category. This is why broiler housing accounts for most of the acidification impacts, though some ammonia evaporates also during cultivation. In terms of tropospheric ozone formation, broiler housing is also the most important phase because of the methane emissions from broiler manure.

According to the LCA results, broiler chicken housing and related fodder production was responsible for most of the global warming potential. Fodder production, especially crop cultivation phases, for broiler chickens accounted for 25% of the primary energy consumed in the production chain, followed by refrigeration in retail stores (20%) and broiler chicken housing (16%). In terms of global warming potential, production of fodder accounted for 36% of the total impact, and broiler housing 29%. This result was not only influenced by the emissions from energy consumption, but also by the nitrous oxide emissions from fertiliser production and use, as well as in nitrous oxide and methane emissions from handling broiler chicken manure. Nevertheless, carbon dioxide remained the most influential greenhouse gas regarding climate change potential, responsible for 59% of all the impacts in that category. Carbon dioxide emissions were evenly distributed throughout the production chain, correlating with the energy consumption. The contribution of the retail trade to climate change was 9% (Fig. 2).

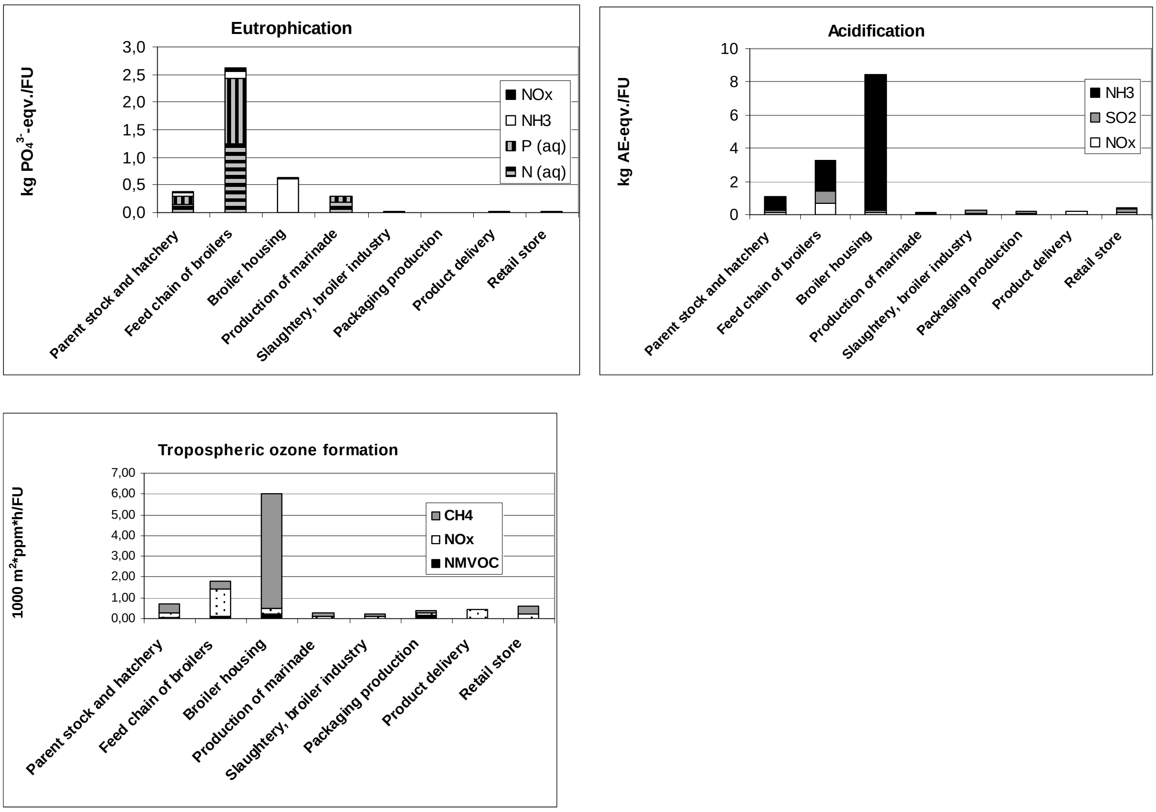

Fig. 1: Eutrophication, acidification and tropospheric ozone formation impact by life cycle phases in the broiler chicken production chain (1000 kg product as FU).

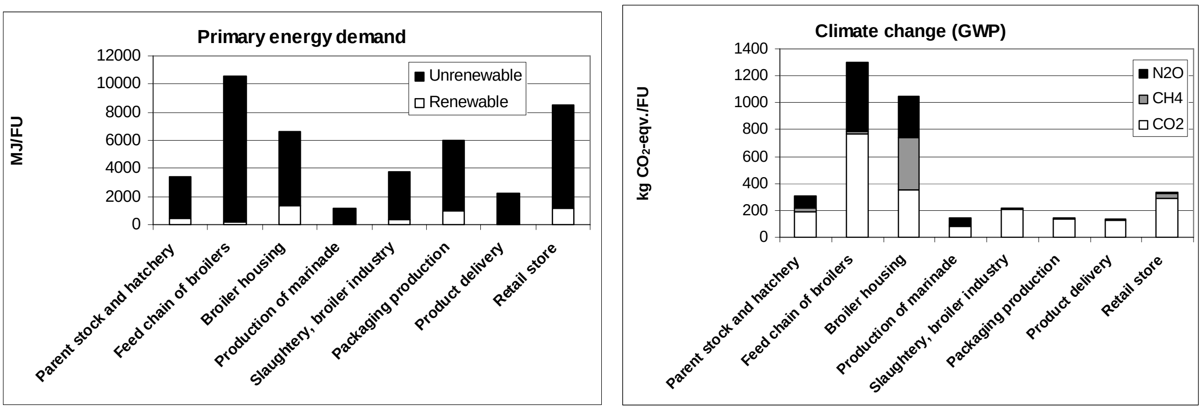

Fig. 2: Primary energy demand and climate change impact by life cycle phases in the broiler chicken production chain (1000 kg product as FU).

3.2 Results illustrated with the Finnish Eco-Benchmark

Using the Eco-Benchmark illustration, the most important phases in the production chain are production of broiler feeds and rearing of broilers. Together these phases accounted for 80% of all environmental impacts. Packaging production, product delivery and retail stores were responsible for 10% of total environmental impacts. The remaining 10% originated from the other phases.

3.3 Improvement options

As the broiler houses seemed to be an important phase in terms of environmental impact, some improvement options designed for that phase were studied. Broiler houses consume considerable energy due to ventilation and heating. There is great potential to save energy using heat recovery systems, but the dust content of the outgoing air, treatment of condensed water and possible hygiene problems represent technical barriers, and for these reasons heat recovery of only 10% efficiency was selected as an improvement option. As a result, savings of more than 33% in heating energy consumption in broiler houses were achieved using this kind of heat recovery and there was a 35% reduction in greenhouse gas emissions in broiler housing (Fig. 3).

With alternative fuels (wood chips and pellets) 70% of greenhouse gas emissions from broiler houses could be cut and even a 6% reduction could be achieved considering the entire production chain. However, this scenario would result in a 7% increase in tropospheric ozone formation due to increased air emissions (Fig. 4).

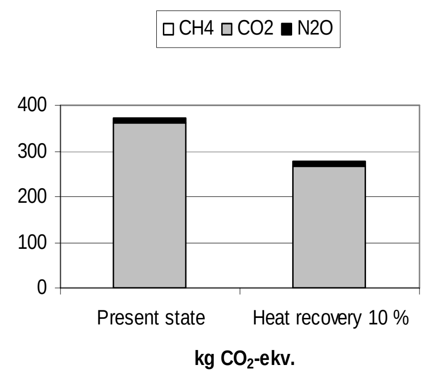

Fig. 3. Change in climate change potential of broiler housing, caused by energy consumption in broiler houses at present state and with heat recovery.

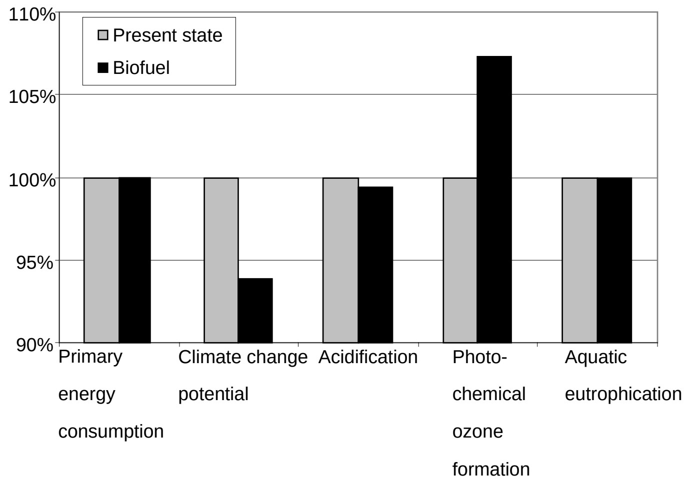

Fig. 4. Change in environmental impacts in "alternative fuel" scenario carried into effect in broiler houses (broiler house heating 50% by wood chips and 50% by wood pellets). Changes in impacts are presented as relative changes (value 100 given to present state).

Reducing ammonia volatilisation in broiler houses is a challenge and it was studied qualitatively. The means to reduce it can be divided into three different groups: a) optimisation of feeds to reduce the nitrogen surplus b) measures to keep the litter dryer and c) cleaning the air in the broiler house. Besides lower ammonia concentration in broiler houses, measures designed to keep the litter dry and in better condition also have other positive impacts in the terms of animal welfare. It is possible to combine ammonia and dust removing processes with heat recovery systems in order to achieve several positive impacts at the same time: lower ammonia and dust concentrations in the broiler house (improved animal health and lower emissions to the atmosphere) and better possibilities for heat recovery (energy saving).

Also some alternative feeding profiles were studied. As a result, using more home-grown grain, a broiler producer would be able to decrease the consumption of primary energy and global warming as the need for transport and feed processing is reduced. Using gas-proof tanks for storing cereals, and avoiding cereal drying, it is possible to save even more energy and reduce greenhouse gas emissions.

However, the more grain from their own fields the broiler producers use, the greater is the total eutrophication impact. The soluble phosphorous in the soil was markedly higher in the broiler farms than in cereal farms due to the long-term use of broiler manure as a fertiliser. As the share of the farms' own grain exceeded 20%, an assumption was made that the field area of the farm is no longer sufficient and grain has to be acquired from surrounding farms.

4. Discussion and conclusions

Crop production for broiler chickens was clearly the most influential component (41%) of the production chain concerning total environmental impacts. The most significant environmental burdens from agriculture were those of nitrogen and phosphorus run-off and leaching (33% of the total impacts by Finnish Eco-Benchmark). The most important target is implementation of new more environmentally sound crop cultivation techniques, both on broiler chicken and feed farms. However, it is much easier to reduce environmental impacts at point-sources, e.g. in broiler houses rather than in cultivation, because control of processes and releases is much more complex under ambient conditions.

Using industrially produced feeds seemed to result in less run-off and leaching than using cereals cultivated on broiler farms. This is due to the high rates of broiler manure applied as fertiliser on broiler farms. This situation can be improved not only by spreading the manure more efficiently among neighbourhood farms, but also by investigating the possibilities to treat the manure industrially. At the same time all means to decrease environmental impacts of agriculture should be brought into play.

In broiler chicken housing, at best the emissions could be reduced by preventing their formation. In this case the quality of litter is important. The litter also plays a significant role in the health of the broilers. In our scenario study, heat recovery proved to be an efficient way to decrease greenhouse gas emissions, but the problem with the equipment is the high investment costs. Decreasing the environmental impacts of the broiler houses should be reviewed as a whole, taking air-conditioning, circumstantial factors and heating into account.

Although farming and broiler production processes in the production chain significantly affected environmental impacts, the shared responsibility in the overall environmental performance of the product has to be recognised widely in the production chain. There is a need to be proactive in cooperating within the entire production chain to find new solutions and to influence collaboration in primary production.

LCA enables the parties involved in the production chain to study their processes and their impacts. The broiler chicken production chain ranges from parent stock to product packaging, and represents a good possibility to develop the entire supply chain. Supply chain integrated LCA furnished participants with new views on cooperation and ideas for modifying the production chain so as to make it more environmentally friendly.

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Environmental hotspot identification of organic egg production

S.E.M. Dekker[1], I.J.M. de Boer[2], A.J.A. Aarnink[3] and P.W.G. Groot Koerkamp[1].
[1] Farm Technology Engineering Group, [2] Animal Production Systems Group, [3] Animal Sciences Group, Wageningen University and Research Centre. Sanne.Dekker@wur.nl

Keywords: life cycle assessment, laying hens, organic eggs, sensitivity analysis

Abstract

According to the ecological principle of the International Federation of Organic Agricultural Movements (IFOAM) "organic farming should be based on living ecological systems and cycles, work with them, emulate them and help sustain them". However, a few ecological problems related to organic egg production are mentioned in literature: 1) depletion of non-renewable energy resources and emission of carbon dioxide caused by long transport distances of concentrates (Bos, 2005) 2) ammonia emission from the laying hen house (Groenestein *et al*., 2005) and 3) eutrophication caused by a high load of nitrogen and phosphorus in the outdoor run, especially in the area close to the hen house (Aarnink *et al*., 2006). Life cycle assessment (Basset-Mens *et al.*, 2006) was used to quantify the relative importance of these problems, identify hotspots and asses the environmental impact of the organic egg production chain. To identify the sensitivity of the LCA outcome to changes in values for production parameters of the laying hen farm, we executed a sensitivity analysis. We chose the baseline impact categories: climate change, eutrophication, acidification, energy use and land occupation. For each impact category, four main clusters were distinguished: i.e. 1) hatching and rearing, 2) concentrate production, 3) egg production and 4) transport. An environmental hotspot was defined as a substance originating from a cluster of processes within the production chain that contributed more than 40% to one of the environmental impact categories**.** Four hotspots were identified. First, 62% of global warming is caused by emission of nitrous oxide from the cluster concentrate production. Second, 57% of acidification is caused by ammonia emission from cluster laying hen farm. Third, 47% of energy use is oil used in the cluster concentrate production and fourth, 95% of the land is used by the cluster concentrate production. From the sensitivity analysis it appeared that the number of eggs produced per hen per year, the feed conversion and the housing system had the largest effect on LCA outcome. An increase in average egg production from 276 by the SD of 39 eggs per laying hen reduced climate change by 13%, acidification by 15%, eutrophication by 13%, energy use by 12% and land occupation by 12%. A reduction in average annual concentrates consumption from 42.9 kg by the SD of 7.2 kg per laying hen reduced climate change by 14%, acidification by 17%, eutrophication by 15%, energy use by 14% and land occupation by 13%. A shift from a single tiered floor housing to multi tiered floor housing with manure drying on belts reduced climate change with 11%, acidification with 53% and eutrophication with 18%. We conclude that for the three mentioned environmental problems only ammonia emission from the hen house is identified as a hotspot. For acidification we conclude that the conversion from a single tiered floor system to a multi tiered floor system with manure drying can be an effective solution. Further on we conclude that concentrate production is the key cluster to climate change, eutrophication and energy use. The laying hen farmer can influence these impact categories by steering on concentrate conversion. However ecologically-sound concentrate production also needs attention.

Introduction

Organic egg production is a fast growing sector in the Netherlands. Between 2005 and 2007, the number of organic laying hens grew from 500.000 to over 900.000. In 2006, 3% of all hens were kept organic, whereas 5.4% of all purchased eggs were produced organic. In 2006, Over 75% of all organic eggs produced in the Netherlands were exported (Biologica, 2007). According to the ecological principle of the IFOAM "organic farming should be based on living ecological systems and cycles, work with them, emulate them and help sustain them" (IFOAM, 2005). So far little research has been done to verify if organic egg production is ecologically-sound, i.e., its environmental emissions and its use of natural resources can be sustained in the long term by the natural environment. However, a few ecological problems related to organic egg production are mentioned in literature: 1) depletion of non-renewable energy resources and emission of carbon dioxide caused by long transport distances of concentrates (Bos, 2005) 2) ammonia emission from the laying hen house (Groenestein *et al.*, 2005) and 3) eutrophication caused by a high load of nitrogen and phosphorus in the outdoor run, especially in the area close to the hen house (Aarnink *et al.*, 2006). The reason experts mention the first problem is that organic hens are fed with concentrate ingredients that originate from all over the world. This is a general tendency in organic farming as shown by world statistics on organic farming (Willer and Yussefi, 2005). Reducing transport by regionalising organic production is mentioned as a possibility to improve ecological sustainability of organic products (Bos, 2005). The second problem is identified by Mollenhorst *et al*. (2006). They concluded in a life cycle assessment (LCA) of conventional egg production that ammonia emission from manure in the hen house of both single and multi-tiered floor systems was the main contributor to acidification. In addition, unlike conventional farmers, organic farmers are not forced by law to build housing systems with low ammonia emissions (VROM, 2004). Regarding the third problem Aarnink (2006) measured nutrient load and ammonia emissions in the first 20 m of two organic outdoor runs and concluded that "…ammonia emission from the outdoor run of laying hens was relatively small compared with the emission from the hen house and that the nutrient load in the outdoor run near the hen house by far exceeded maximum acceptable levels." The relative importance of the above described three problems concerning organic egg production is currently unknown. To gain insight into ecological sustainability of organic egg production, the environmental impact of organic egg production should be assessed in an integral way. Integral, in this respect, means incorporation of all relevant environmental impacts and all processes involved in the production of organic eggs. LCA is a widely accepted method for integrated environmental impact assessment of food products. The aim of this research, therefore, is to quantify the integral environmental impact of the organic egg production chain using LCA in The Netherlands. Such an assessment can also reveal the environmental hotspots in the egg production chain. In addition, to identify powerful production parameters on the laying hen farm a sensitivity analysis will be done.

Material and Methods

The four stages of an LCA, i.e. goal and scope definition, inventory analysis, impact assessment and interpretation of results, are described below.

Goal and scope definition

To evaluate the integral environmental impact of the organic egg production chain, we used attributional LCA (Thomassen *et al.*, 2008). The functional unit was defined as one kg of organic egg leaving the farm gate. In accordance with Guinée *et al*. (2001) we chose the baseline impact categories: climate change, eutrophication, acidification, energy use and land occupation. The selection of these impact categories depended on the availability of data and on their relevance for animal production. The system boundaries, as visualised in Fig. 1, included the processes: cultivation of concentrate ingredients, transport to the concentrate factory, concentrate processing, transport of concentrates to the farm, hatching of eggs, transport of the hatcher to the rearing farm, rearing of the hen, transport of the reared hen to the laying hen farm and egg production on the laying hen farm. The environmental impact of a process with several co-products was allocated based on the relative economic value of the products. Processes needing allocation were production of concentrate ingredients and their co-products and production of eggs, slaughter hens and manure. From data collection it was concluded that the economic value of organic laying hen manure was zero. Production of buildings, medicines and machinery, except transport lorries and litter were excluded from the LCA.

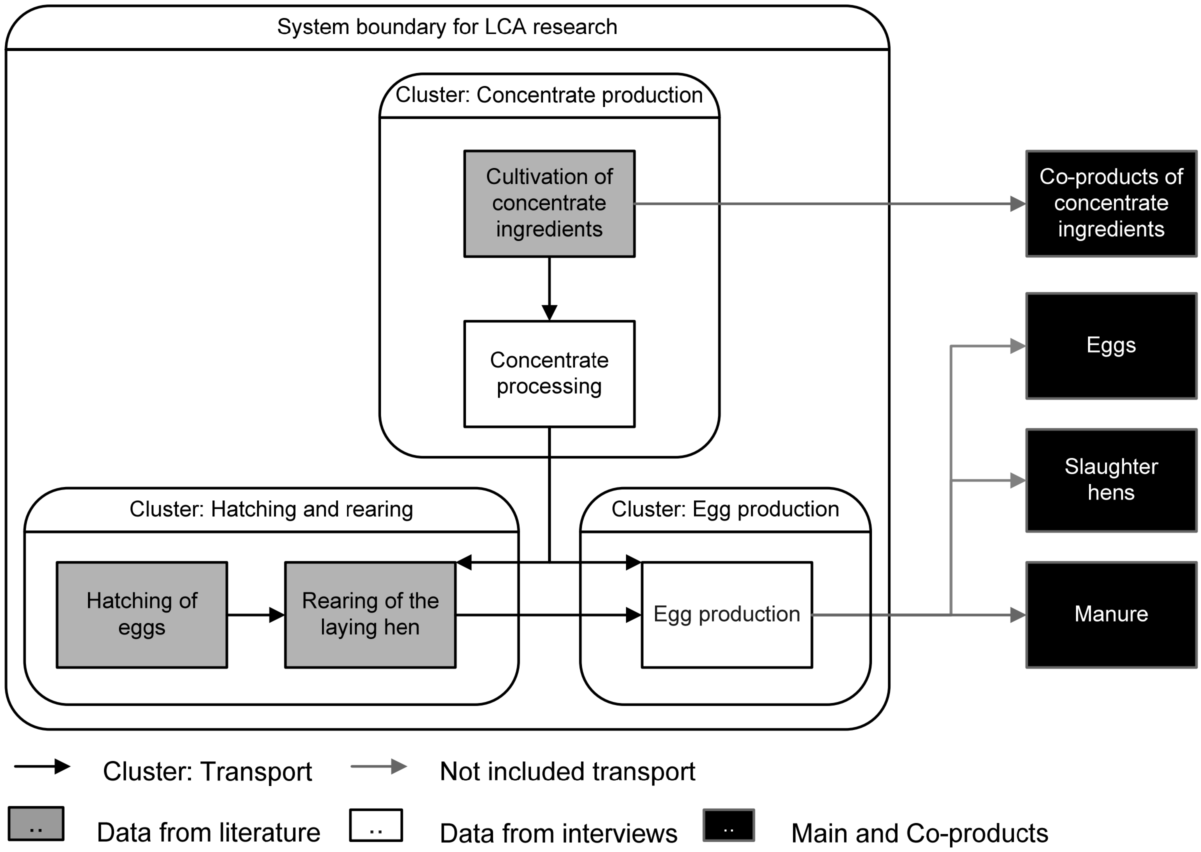

Fig. 1: System boundary of the LCA study, included processes, included and excluded transport, main product and co-products, data origin and the division in four clusters as used for hotspot identification.

Inventory analysis

The inventory analysis consisted of collection of data concerning relevant inputs, outputs and environmental losses for each included process (see Fig. 1). Based on this information, a life cycle inventory (LCI) for each process was computed. Required data were collected from literature, interviews with egg producers and feed industries. Below, the LCI of the main processes is described more in detail.

Hatching of eggs

Required production parameters for the hatching process were based on average statistics of conventional hatcheries (ASG-WUR, 2004; Hemmer *et al.*, 2006). Information was gathered about 1) inputs of the hatching process, e.g., number of hatching eggs, the use of electricity, water, natural gas, methanol, formaldehyde and land and 2) output of the hatching process i.e., the number of hatchers produced. Hatching of one egg requires 0.92 kWh electricity, 1.65×10^{-5} formaldehyde, 3.12×10^{-6} methanol, 0.92 l water, 0.18 MJ natural gas and 1.8×10^{-4} m^2 land. Mortality, including selection, was 60% and hatching time was 25 days. Production of hatching eggs by laying hen breeders was neglected since we calculated that this process would contribute less than 1% to the LCA of the organic egg.

Rearing of the laying hen

Rearing hen farms produce conform regulation EC 2092/91 for organic farming (EG, 1991) and Skal (2009). We, however, used average statistics from conventional rearing hen farms (Hemmer *et al.*, 2006) and regulations to estimate production parameters of organic rearing hens (see Tab. 1). Information was gathered about: 1) farm inputs, i.e. number of hatchers and amount of concentrates, 2) farm outputs, i.e. reared hens and 3) on-farm data required to calculate environmental impacts, i.e.

land occupation, housing system and manure excretion in the outdoor run. The use of electricity, gas, diesel, water and litter was not included in this research due to lack of data.

Egg production

Laying hen farms produce conform regulation EC 2092/91 for organic farming (EG, 1991). We conducted interviews with 20 out of a total of 68 Dutch organic laying hen farmers to collect data on their last production round finished in 2006 or 2007. Farms were selected randomly from a complete database including all Dutch organic laying hen farms in the Netherlands with over 1500 laying hens. Approximately half of the contacted farmers participated in the conduction of the interview. Information was gathered about 1) farm inputs, i.e. reared hens, purchased concentrates and wheat; 2) farm outputs, i.e. eggs, slaughter hens and manure and 3) on-farm data required to calculate environmental impacts, i.e. land occupation, housing system and manure excretion in the outdoor run. It was concluded from the interviews that farmers did not use gas and diesel. The use of electricity, water and litter was not included due to lack of data. Tab. 1 shows the mean and corresponding standard deviation of production parameters of 20 laying hen farms and values assumed for the rearing hen farm. The means of the 20 interviews were used as input parameters in the LCA model. Data on the emission of ammonia, nitrous oxide, nitrogen oxides and methane from the hen house and outdoor run and eutrophication caused by phosphorus and nitrate from the outdoor run of all farm types were based on literature for conventional farming (Oenema *et al*., 2000). Subsequently these data were modelled according to Groenestein (2005) to depend on the following farm characteristics: housing system, storage time, total nitrogen and phosphorus excretion and manure excretion in the outdoor run.

Tab. 1: Production parameters of the organic laying hen farm and organic rearing hen farm per production round.

Production parameter	**Unit**	**Laying hen farm**	**Rearing hen farm**
		Mean (SD)	Mean
Purchased hens	hen/farm	7604 (4281)	-
Single tiered floor housing	%	85	100
Multi tiered floor housing	%	15	0
Stocking density house	hen/m^2 [b]	5.55 (0.78)	5.55
Stocking density outdoor run	hen/m^2 [b]	0.22 (0.04)	1
Length of round	days	398 (44)	119
Purchased concentrates	kg/hen [b]	43(7)	6
Purchased wheat	kg/hen [b]	4.6	0.6
Hens in outdoor run [a]	%	9 (4)	5
Mortality rate	%	13 (5)	3.9
Egg weight	g	63 (2)	-
Egg production	#/hen [b]	276 (39)	-
Egg price	euro/kg	1.83 (0.2)	-
Start weight hen	kg	1.52	0.035
End weight hen	kg	1.94 (0.09)	1.52
Slaughter price	euro/kg	0.18 (0.12)	-
N-excretion	kg N/hen [b]	0.96 [c]	0.11
P-excretion	kg P/hen [b]	0.20 [c]	0.02

[a] Average amount of hens the farmer estimated to be present in the outdoor run during the day.
[b] Amount of reared hens the farmer purchased.
[c] No SD available because N- and P-excretions are LCA model output values.

Transport

According to the interviewed farmers and concentrate industries resources were transported by lorry or transoceanic freight ship. The LCI for transportation with a transport lorry with a maximum transportation load of 32 tonnes and transoceanic freight ship originated from Ecoinvent V 2.1

database (Ecoinvent Center, 2008) and were expressed in kg transported product per km. From the interviews we estimated that the transport distance of a rearing hen was 50 km, of a hatcher 99 km and of concentrates from the concentrate industry to the farmer 50 km. Tab. 4 contains transport distances for various concentrate ingredients. Packaging material was not incorporated into the computation of transport weight. The weight of a hatcher was assumed to be 35 g and of a rearing hen 1.52 kg (Jongbloed and Kemme, 2005). The interviews with laying hen farmers showed that 50% of the farmers purchased wheat from the concentrate industry and 50% from their own region. For the former, the total transport distance of wheat from the arable farmer via the concentrate industry to the laying hen farmer was 595 km. For the latter, the transport distance of wheat was 10 km.

Cultivation of concentrate ingredients and wheat and processing of concentrates

Concentrates were produced conform regulation EC 2092/91 (EG, 1991). We conducted interviews with two out of four Dutch industries that produce concentrates for organic hens, to collect data on their concentrate production in 2007. Information was gathered about 1) the inputs, i.e. characterisation of concentrate ingredients, concentrate composition and the amount of electricity, diesel, water and gas required for production; 2) the output, i.e. the amount of concentrates produced and 3) industrial data required to calculate the environmental impact, i.e. land occupation of the factory. It was concluded from the interviews that the production of concentrates for organic laying hens needed hardly any water, gas and diesel, because concentrates were not pelleted. Per tonne concentrate 7 kWh electricity and 0.0009 m^3 of factory land was used. Due to practical reasons the concentrate composition was simplified in the LCA model into one concentrate type for rearing as well as laying hens and 8 concentrate ingredients. The organic status of the concentrate ingredients and the modelled concentrate composition are specified in Tab. 2. The production of wheat involved no processing. For the cultivation of concentrate ingredients and wheat information was gathered about 1) farm inputs, i.e. seed, diesel and electricity 2) farm outputs, i.e. kg dried and processed concentrate ingredient and straw 3) on-farm data required to calculate the environmental impacts, i.e. land occupation, emission of ammonia, nitrous oxide and eutrophication caused by phosphorus and nitrate. Potatoes, soya beans, wheat and maize are dried after yielding. Economically allocated co-products were straw, soy oil, and sunflower oil. Concentrate composition and cultivation characteristics of the 8 concentrate ingredients are visualised in Tab. 2. The LCI of monocalcium phosphate was available from the Ecoinvent V2.1 database (Ecoinvent Center, 2008). Data on cultivation and processing of the ingredients were derived from Dekkers (2002) and Thomassen (2008).

Tab. 2: Composition of organic concentrates for rearing hens and laying hens, origin of the ingredients, specification on organic or non organic status, transportation distance, seed use, diesel use electricity use and allocation percentage.

Ingredient	**Origin**	**EKO**	**Share**	**Transp.**	**Yield**[a]	**Seed**	**Diesel**	**Electr.**	**Alloc.**
	country	Y/N	%	km	kg/ha	kg/ha	l/ha	MJ/ha	%
Maize	IT	Y	31.9	250	4938	150	180	-	100
Wheat	DE, IT, RU	Y	33.5	545	4125	200	106	52	89
Sunflower seed Expeller	NL, EU	Y	3.8	500	1121	200	100	177	36
Potato protein	NL	N	4.3	250	5208	2300	227	220	100
Peas	DE, IT	Y	6.0	250	4250	150	71	76	100
Alfalfa	NL	Y	2.8	250	12000	25	23	-	100
Soya bean expeller	BR	Y	8.3	10845	1762	200	106	125	72
Monocalcium Phosphate	BE	N	5.6	500	-	-	-	-	100

[a] The yield is expressed in kg dried and processed concentrate ingredient.

Impact assessment

Relevant substances per impact category were selected based on knowledge from earlier LCA studies on animal products and are listed in Tab. 3. Next to the calculation of the total environmental impact per kg of egg, we also identified environmental hotspots for each environmental impact category. An environmental hotspot was defined as a substance originating from a cluster of processes within the production chain that contributed more than 40% to one of the environmental impact categories**.** Results are presented for each impact category separately. For each impact category, four main clusters were distinguished: i.e. 1) hatching and rearing, 2) concentrate production, 3) egg production and 4) transport (see Fig. 1). To identify the sensitivity of the LCA outcome to changes in values for production parameters of the laying hen farm, we executed a sensitivity analysis. This sensitivity analysis implied that we examined the effect of a value change of production parameters, such as the number of eggs produced per hen per year or the feed conversion, on final LCA results. For continuous parameters, we explored the effect of a positive or negative deviation of one standard deviation from the mean, whereas for discontinuous parameters, such as housing system, we compared single-tiered floor housing and multi-tiered housing with manure belts and manure drying.

Tab. 3: Selected impact categories with related units, contributing elements, characterisation factors and references (IPCC, 2006).

Impact category	**Unit**	**Contributing elements**	**Characterization factors**
Climate change	kg CO_2 eq	CO_2	1
		CH_4	23
		N_2O	296
Acidification	kg SO_2 eq	SO_2	1
		SO_x [a]	1.2
		NH_3	1.88
		NO_x	0.7
Eutrophication	kg PO_4^{3-} eq	PO_4^{3-}	1
		P_2O_5	1.34
		H_3PO4	0.97
		P	3.06
		NH_3	0.35
		NH_4^+	0.33
		NO_x	0.13
		NO_3^-	0.1
		N	0.42
Energy use	MJ LHV/kg	oil	41-45.8
		gas	30.3-49.8
		uranium	451000-2291000
		coal	8-29.3
Land occupation	m^2	land occupation	1

[a] SO_2 as SO_x

Results

An overview of LCA results is given in Tab. 4. We identified four environmental hotspots. First, 62% of global warming is caused by emission of nitrous oxide during production of concentrates. Second, 57% of acidification is caused by ammonia emission on the laying hen farm. Third, 47% of energy use is oil used for production of concentrates and fourth, 95% of the land occupation is required for production of concentrates. We identified no hotspot for eutrophication, but production of concentrates contributed most with 37% nitrogen leaching and 26% phosphate accumulation.

Tab. 4: Preliminary results of the environmental impact assessment in g equivalent per kg organic egg for the environmental impact categories: climate change, acidification, eutrophication, energy use and land occupation.

Climate change (g CO_2-eq./kg) egg)	**N_2O**	**CO_2**	**CH_4**	***Total***	
Egg production	548	40	71	*659*	
Rearing and hatching	65	9	27	*102*	
Concentrate production	2475[a]	534	12	*3020*	
Transport	3	248	7	*258*	
Total	*3090*	*831*	*117*	*4038*	
Acidification (g SO_2-eq./kg egg)	**NH_3**	**NO_x**	**SO_x**	***Total***	
Egg production	45.7[a]	2.1	0.0	*47.8*	
Rearing and hatching	5.8	0.3	0.0	*6.0*	
Concentrate production	17.8	1.9	4.5	*24.2*	
Transport	0.0	1.6	0.3	*1.8*	
Total	*69.3*	*5.8*	*4.8*	*79.9*	
Eutrophication (g PO_4^--eq./kg egg)	**N-water**	**N-air**	**PO_4^-**	***Total***	
Egg production	0.0	8.9	2.0	*10.9*	
Rearing and hatching	0.0	0.0	0.0	*0.0*	
Concentrate production	14.4	3.7	10.2	*28.3*	
Transport	0.0	0.3	0.0	*0.3*	
Total	*14.4*	*12.9*	*12.2*	*39.5*	
Energy use (MJ/kg egg)	**Oil**	**Gas**	**Uranium**	**Coal**	***Total***
Egg production	0.0	0.3	0.0	0.2	*0.6*
Rearing and hatching	0.0	0.1	0.0	0.0	*0.1*
Concentrate production	5.4[a]	1.3	0.8	0.7	*8.1*
Transport	3.5	0.3	0.2	0.2	*4.3*
Total	*9.0*	*1.9*	*1.1*	*1.1*	*13.1*
Land occupation (m^2/kg egg)	***Total***				
Egg production	*0.3*				
Rearing and hatching	*0.0*				
Concentrate production	*6.1*[a]				
Transport	*0.0*				
Total	*6.4*				

[a] Identified as hotspot because value contributes more than 40% to total of environmental impact category.

From the sensitivity analysis it appeared that the number of eggs produced per hen per year, the feed conversion and the housing system had the largest effect on LCA outcome. An increase in annual egg production per hen from 276 eggs with a SD of 39 eggs reduced climate change with 13%, acidification with 15%, eutrophication with 13%, energy use with 12% and land occupation with 12%. An improvement of the feed conversion by reducing the average annual concentrates consumption from 42.9 kg with the SD of 7.2 kg per laying hen reduced climate change with 14%, acidification with 17%, eutrophication with 15%, energy use with 14% and land occupation with 13%. A shift from single-tired floor housing to multi-tired housing with manure drying reduced climate change with 11%, acidification with 53%, eutrophication with 18% and had no effect on land occupation.

Tab. 5: Reduced percentage of the LCA in three different scenarios; 1) an increase of egg production by its SD based on the laying hen farm interviews, 2) a reduction of concentrate consumption by its SD based on the laying hen farm interviews, 3) A shift from a single to a multi tiered housing system with manure drying on manure belts

Production parameter	**Egg production**	**Concentrate consumption**	**Housing**
	#/hen*year	kg/hen	Type
Current situation	276	42.9	100% single tiered
Scenario	+39	-7	100% multi tiered
Impact category (unit)			
Climate change (g CO2-eq./kg egg)	-13%[a]	-14%[a]	-11%[b]
Acidification (g SO2-eq./kg egg)	-15%[a]	-17%[a]	-53%[b]
Eutrophication (g $PO4^-$-eq./kg egg)	-13%[a]	-15%[a]	-18% [b]
Energy use (MJ eq. /kg egg)	-12%[a]	-14%[a]	-?[c]
Land occupation (m^2 eq./kg egg)	-12%[a]	-13%[a]	0%[a]

[a] Reduction of LCA compared to originally calculated model.
[b] Reduction compared to 100% single tiered floor scenario.
[c] No representative results available, since no data were available on differences in energy use between single tiered and multi tiered housing systems.

Discussion

Regarding the three environmental problems described in the introduction we concluded that 1) transport is not identified as a hotspot, but contributed 33% to total energy use 2) ammonia emission from the hen house was identified as a hotspot and 3) potential environmental impact of manure deposition in the outdoor run (i.e. eutrophication) on the laying hen farm was not identified as a hotspot. This LCA study, however, could be extended by including litter, machinery, buildings, energy and water use and distribution of the egg. The sensitivity analysis should be extended to all parameters of the production chain. Also the uncertainty of the system caused by model parameters should be analysed.

Conclusion

We conclude that for the three mentioned environmental problems only ammonia emission from the hen house is identified as a hotspot. For acidification we conclude that the conversion from a single tiered floor system to a multi tiered floor system with manure drying can be an effective solution. Further on we conclude that concentrate production is the key cluster to climate change, eutrophication and energy use. The laying hen farmer can influence these impact categories by steering on concentrate conversion. However ecologically-sound concentrate production also needs attention.

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Environmental Impacts of Alternative Uses of Rice Husks for Thailand

J. Prasara-A[1], T. Grant[2]
[1]School of Global Studies, Social Science and Planning, RMIT University, Melbourne, Australia, jittima.prasara-a@rmit.edu.au

[2]Centre for Design, RMIT University, Melbourne, Australia

Keywords: Bioenergy, agricultural waste, system expansion, rice husk

Abstract

This study compares the environmental impacts of different rice husk use pathways, i.e. use in power generation, cement manufacture and cellulosic ethanol production. A consequential LCA method has been used in the comparison of options to determine how these beneficial uses of rice husks will lead to substitution of virgin materials such as fossil fuel, cement raw materials and petroleum product, and changes in the emission profiles of these production systems. As a result, compared to the conventional systems such as the Thai grid production, ordinary Portland cement and petrol production, using rice husks in the three systems investigated cause lower impacts on fossil fuels consumption and climate change. However, the impact on other indicators investigated is higher than that of those conventional production systems. The most favourable option for disposal of the rice husk ash produced from power generating production is using it in light weight concrete block production as it causes less impact on all indicators analyzed. The most environmentally favourable rice husks use system with regard to fossil fuels consumption is the use in power generation compared with the use in cement manufacture and cellulosic ethanol production. The cement manufacture system is the most preferable when climate change is considered.

Introduction

Thailand is one of the largest rice producing countries in the world. In recent year, the nation produces about 29 million tonnes annually (Office of Agricultural Economics 2006). Rice husks, which are a by-product of rice production, account for 23% of total paddy weight. Being light and bulky, the husks cause significant disposal problems for the rice mill owners. Furthermore, the methane gas that is released when the husk is fermented by micro-organisms can contribute to global warming. Also, the rice husk is one of the potential biomass sources in Thailand. The Thai government has encouraged the use of biomass fuel to help reduce global climate change and reserve fossil fuel resources. Therefore, rice husks have been utilized in several ways.

One of the ordinary uses of rice husks in Thailand has been as a source of energy within the rice mills. However, there were still surplus rice husks from the process after being used in paddy drying and milling (The EC-ASEAN COGEN Programme 1998). Also, there have been some minor uses such as using in livestock farms, farmland, charcoal production and brick production, etc. More recently, rice husks have been put to use within the industrial sectors such as electricity generation and cement manufacture as an energy source.

Rice husks as one of cellulosic materials can be used as a feedstock in the Cellulosic ethanol production. There has been intensive research on converting lignocellulosic biomass into ethanol and much effort has been put to introduce it on a large scale manufacture in other countries like USA, Canada and some European countries (Hahn-Hägerdal *et al*. 2006; Lin & Tanaka 2006; Saha *et al*. 2005). Even though this technology has not yet been introduced to Thailand, it is one of the potential uses of rice husks when the technology has been proven.

Although there seem to be several alternative ways of disposing of rice husks, the environmental impacts of these potential systems have not yet been widely investigated within the Thai context.

This study assesses the environmental impacts of the selected main beneficial uses of rice husks, i.e. use in power generation, cement manufacture and cellulosic ethanol production, and determines whether the use of rice husks in those systems investigated will lead to reducing environmental impacts compared to those conventional processes.

Method / Approach

A consequential LCA approach was taken in this study. Since the aim of this study is to indicate the favourable use of rice husks from the environmental point of view, a consequential LCA is considered appropriate for the study. There has been a discussion about the proper LCA approach to perform, whether it should be attributional or consequential LCA, though sometimes different terms like retrospective and prospective LCAs have been used instead. It is suggested that if the reason for conducting an LCA is for decision making support purposes, then a consequential LCA method is more appropriate (Ekvall, Tillman & Molander 2005; Ekvall & Weidema 2004; Tillman 2000)

Goal and Scope of the Study

The goals of this study are to assess the environmental impacts of different rice husks use pathways; and to determine whether the use of rice husks in different product systems will result in reducing the environmental impacts compared to the conventional systems. Three main rice husks alternative uses selected to be examined are the use in power generation, cement manufacture and cellulosic ethanol production.

The systems investigated include only processes that are affected by changes in the systems analyzed; the rice production is excluded from the study (a system boundary is shown in Fig. 1). This consequential LCA avoids co-product allocation by means of a system expansion. In some rice husks use options, there is a co-product generated in the same process, this is called multifunctional process. To deal with this allocation problem, it is suggested that the co-product allocation is avoided by system expansion (Ekvall & Weidema 2004; Weidema 2001). Assuming that co-products are fully utilized, models describing system expansion for each option are shown in Fig. 2 and 3.

In the power generation process, rice husk ash is produced from the rice husks combustion process. This ash is sent to other ash consumers such as soil conditioner, clay brick and lightweight concrete block production. These are taken into account for the model. However, the model does not include the whole production processes of these products. Production processes of the competing products of rice husk ash like chemical fertilizer, clay, Portland cement are avoided in this model.

For the cellulosic ethanol process, there is also a co-product generated in the process. There are the solid residues left out from the ethanol process, which consists of mainly lignin from rice husks. These residues are burned in cogeneration plants to produce both steam and electricity to use in the ethanol plant itself. It is assumed that both heat and electricity produced from the cogeneration plant are enough for internal use and excess electricity is sold to the grid. This gives environmental credit to the cellulosic ethanol production; therefore, an amount of electricity sold to the grid is avoided in this model.

In the cement production process, rice husks are used as an energy source to substitute for coal. The husks are burnt to produce heat in the clinker burning process. Its ash is mixed with clinker to produce cement, this means that rice husk ash finally comes out as part of the cement product. Therefore, system expansion is not applied for this model since there is no co-product in this process.

Functional units given for each system are different depending on the production that rice husks are employed in. They are defined as 1 MWh for the power generation system, 1 tonne for cement production and 1 kg for cellulosic ethanol system. However, the results presented in this paper refer to the environmental impacts caused by consuming 1000 tonne of rice husks in each system.

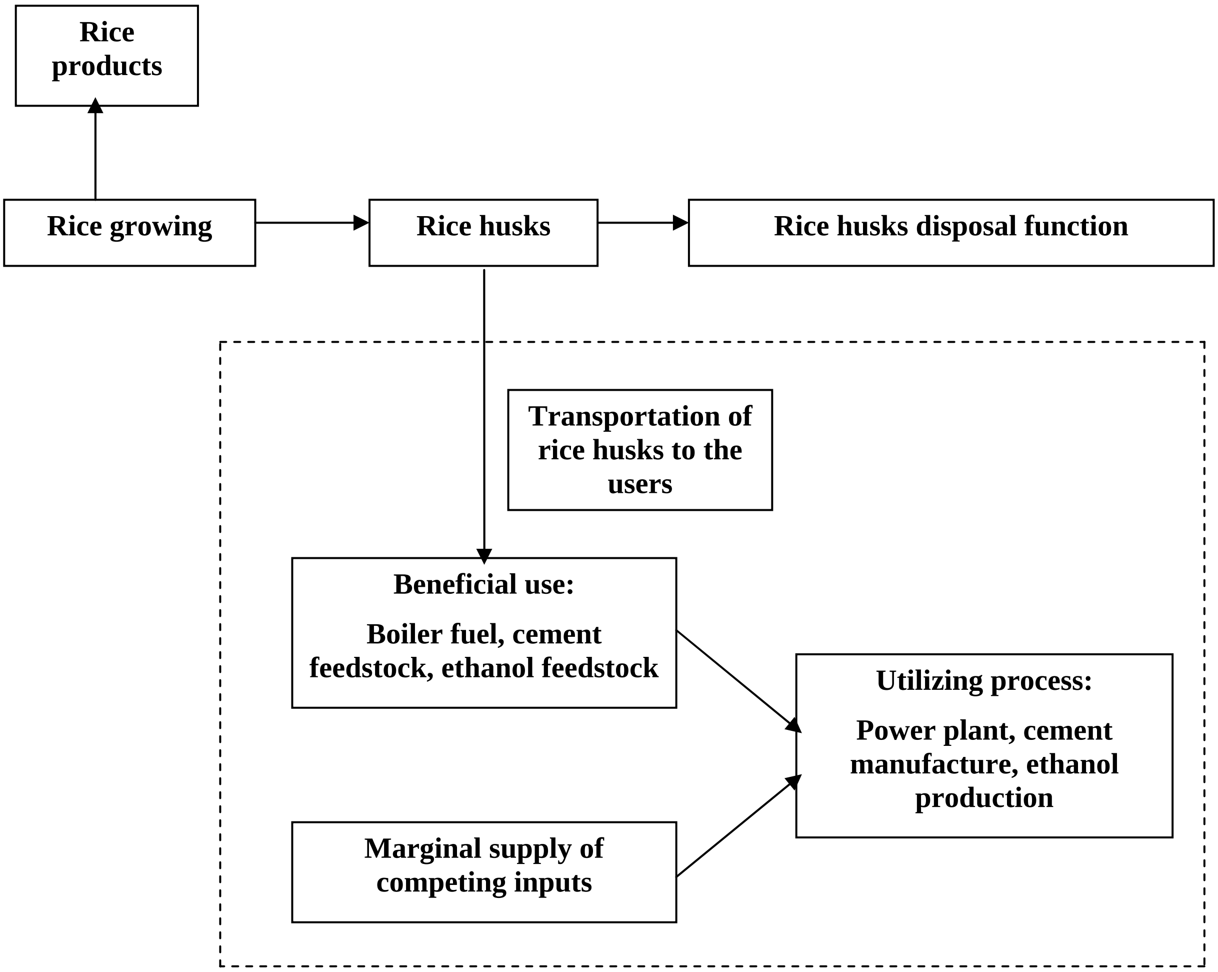

Fig. 1: System boundary of the study

Inventory Analysis

Foreground data are obtained from interviews with industry personnel, LCA questionnaires and literature. Background data are from LCI databases available (Ecoinvent, Australian Life Cycle Inventory Database) and literature. It is worthy to note that the Thai LCI database is developing and it has not yet been made available to the public. However, LCI data for some production processes are available from published reports (Lohsomboon & Jirajariyavech 2003; Thailand Environment Institute (TEI) 2004).

LCI data for the power generation option are mainly collected from one specific rice husk power plant. However, some data are taken from other literature sources to close data gaps.

LCI data for the Thai cement production are based on the report by Thailand Environment Institute (TEI) (2004). This report shows LCI data for conventional Thai Portland cement production for which rice husks were not included in the process. The LCI data for production process of Portland cement with rice husks replacing 20 % of coal are adapted from the exiting data in this report.

This was done based on an assumption that rice husks are used to substitute for coal by 20% concerning energy content and that coal ash and rice husk ash have fairly the same chemical compositions. Al_2O_3 and Fe_2O_3 contained in rice husk ash are not taken into account because they are very small amounts. After substituting rice husks into the cement process, some part of shale are taken out since rice husk ash contributes SiO_2 to the clinker (shale is the main raw material providing SiO_2 into the process) However, shale also provides Al_2O_3 and Fe_2O_3 to the process. Hence, Al_2O_3 and Fe_2O_3 are lacking as a result of having removed some shale from the process. These chemicals are reintroduced when bauxite and Iron ore are added to the process (TEI 2004). Due to limited data, all emissions are assumed to be the same as the conventional Portland cement production (i.e. without rice husks). However, the fossil CO_2 amount is deducted based on a calculation of CO_2 emitted by burning the amount of coal replaced by rice husks.

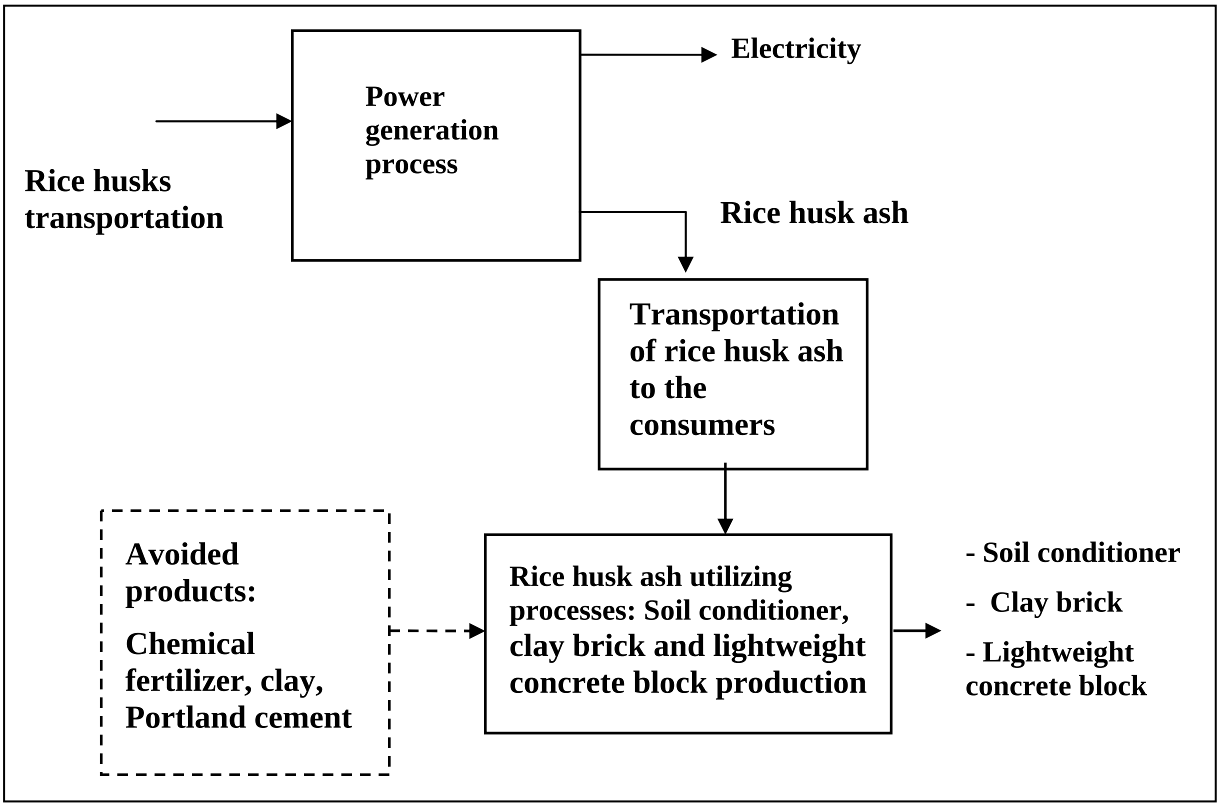

Fig. 2: LCI Model describing system expansion for Power generation option

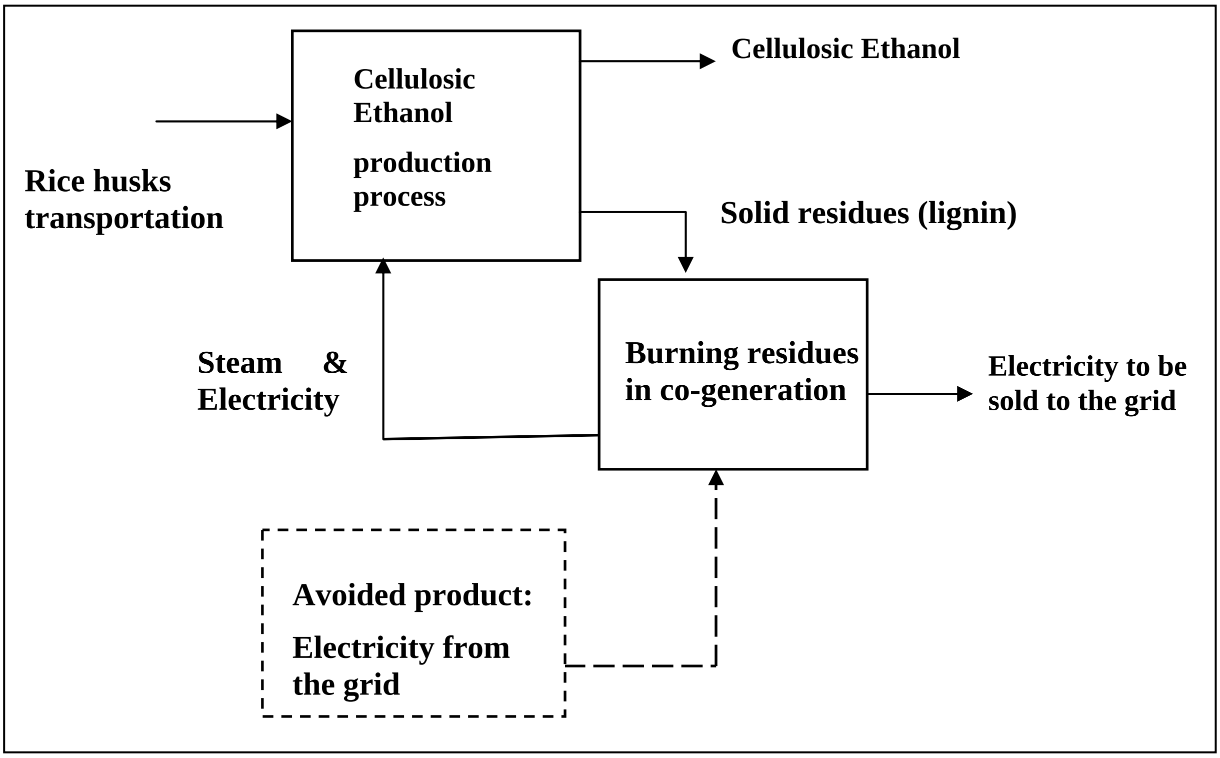

Fig. 3: LCI Model describing system expansion for Cellulosic ethanol option

As the cellulosic ethanol production has not yet been introduced to Thailand, data from other countries are used in this study. There are no LCI data for cellulosic ethanol from rice husks available; the data used for this model are adapted from the production process of cellulosic ethanol production from wood (Jungbluth *et al*. 2007). Specifically, ethanol yield is adjusted to rice husks conditions according to Saha *et al*. (2005). While inputs from technosphere are proportional to dry matter input. Emissions of hydrocarbons are proportional to carbon input and all other emissions are proportional to dry matter input (Jungbluth *et al*. 2007).

Some process data used in this study are adapted for the Thai conditions where possible. However, there were only a few sets of LCI data of the Thai production processes available, most data employed in this work are based on unit process data from Australia and European countries.

With regard to the consistency of data, all unit process data within SimaPro7 used in this study are adapted to the same detail level. For instance, LCI data for some processes include infrastructure and waste management processes but some do not. In this study, infrastructure and waste management processes are taken off to make the system models investigated comparable. Based on carbon neutral concept, CO_2 released by burning rice husks is accounted as neutral and this credit was given to all rice husks use systems to make them comparable.

Impact Assessment

The impact assessment method used in this study is Eco-indicator 99 (H) V2.05 / Europe EI 99 H/H / normalization. It was used in the way that it was set up in SimaPro7. The impact indicators analyzed were abiotic depletion, global warming, human toxicity, ecotoxicity, photochemical oxidation, acidification, eutrophication.

Results and Discussion

For the electricity production system (see Fig. 4), it shows that using rice husks to generate electricity causes lower impacts on fossil fuels consumption and climate change**.** Compared to the Thai grid production, using rice husks to generate power causes little impact on fossil fuels consumption since there is only little amount fossil fuels needed in the transportation of rice husks from the rice mill. As energy from biomass, generating power from rice husks does not contribute to climate change. However, the impact on respiratory inorganics appears higher compared to the Thai grid production. This may result from higher particulate matter produced when burning rice husks to generate electricity as also discussed in a previous study (Chungsangunsit 2004).

The most environmentally preferable option for a disposal of rice husk ash produced from the rice husk power plant is the use of the ash in light weight concrete block production. This option causes a little less impact on respiratory inorganics, climate change and fossil fuels. All other uses have similar benefits. This results from the higher credit given to this rice husk ash disposal option by substituting rice husk ash for Portland cement in the concrete block production process.

Fig. 5 shows a comparison of the normalized impacts that were reduced by consuming 1000 tonnes of rice husks in the different three systems. For the electricity generation system, the option of sending rice husk ash from the rice husk power plant to the light weight concrete block production plant is taken into this comparison as it is the most preferable rice husk ash disposal option. As a result, it is shown that using rice husks in power plants has the greatest effect in reducing impact on fossil fuels, followed by ethanol and cement manufacture respectively. With regard to climate change, the cement option scores better than others, and cellulosic ethanol option seems to provide minimal help to reduce the impacts compared to the conventional processes.

Compared to the conventional Portland cement production, using rice husks in the cement manufacture process is better for climate change and fossil fuels consumption indicators. This results from using rice husks to substitute some part of coal in the cement production process so this also helps to reduce green house gases from burning coal.

The cellulosic ethanol option is obviously better than petrol production with respect to fossil fuels consumption. With regard to climate change, the cellulosic ethanol option seems to offer very little benefit compared with the petrol production. However, the petrol production is better in the effect on respiratory inorganics. These may result from the process of burning solid residues left from ethanol distillery to produce heat and electricity for use in the ethanol plants and then sell the surplus amount to the grid, and also from the production process of sulphuric acid as making cellulosic ethanol from rice husks requires more sulphuric acid compared to other lignocellulosic material such as wood (Jungbluth *et al*. 2007; Saha *et al*. 2005). However, the work described in this paper only analyses the impacts of the cellulosic ethanol production compared to the petrol production. It does not include an analysis of impacts caused by using the cellulosic ethanol produced from rice husks compared with the use of petrol in vehicles. This should be further investigated.

In general, all rice husk use systems analyzed are all better in terms of fossil fuels consumption and climate change. Nevertheless, they are not better than the conventional processes concerning other impact indicators evaluated.

A comparison of the weighted impacts reduced by consuming 1000 tonnes of rice husks in the different three systems is shown in Fig. 6. These results show that the electricity option gives the largest benefit over the conventional process, along with ethanol and cement options in resources category; as discussed earlier it has the largest effect in reducing fossil fuels consumption. In the human health category, the cement option seems to provide the most benefit compared with the other options as it helps to reduce green house gases by the largest amount, and ethanol is the worse in this damage category as it causes higher impacts in respiratory inorganics compared with the conventional process. In addition, the ethanol option is a little unfavourable in ecosystem quality since the production process of ethanol from rice husks causes little higher impacts in ecotoxicity, acidification and eutrophication when compared to the petrol production. However, these weighted results have to be interpreted carefully as they are subjective.

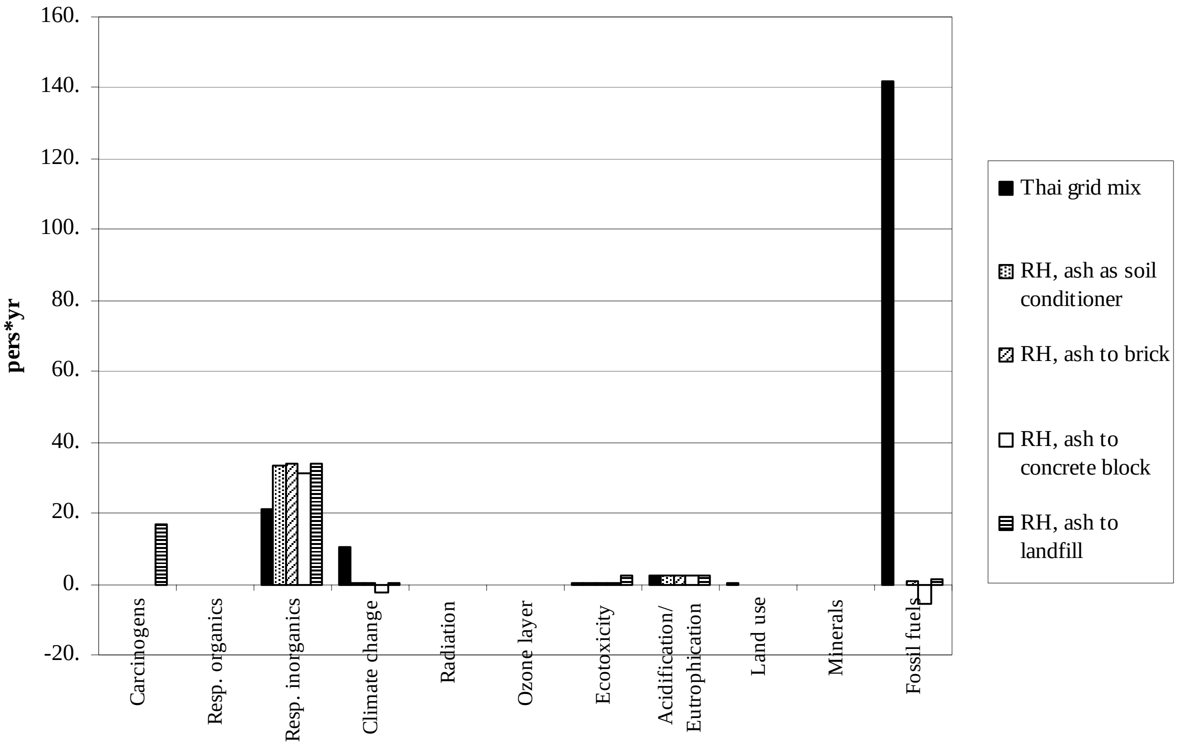

Fig. 4: A comparison of normalized impacts of rice husk power plant production with the production of Thai grid

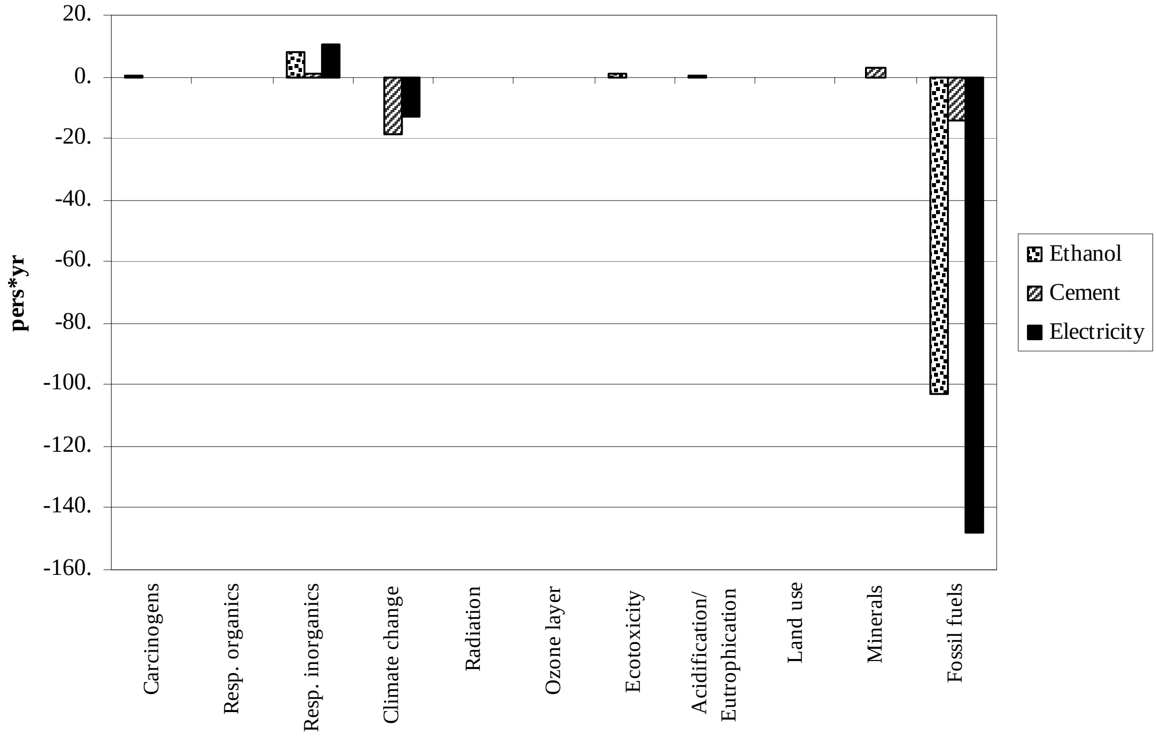

Fig. 5: A comparison of normalized impacts reduced by consuming 1000 tonnes of rice husks in different use systems

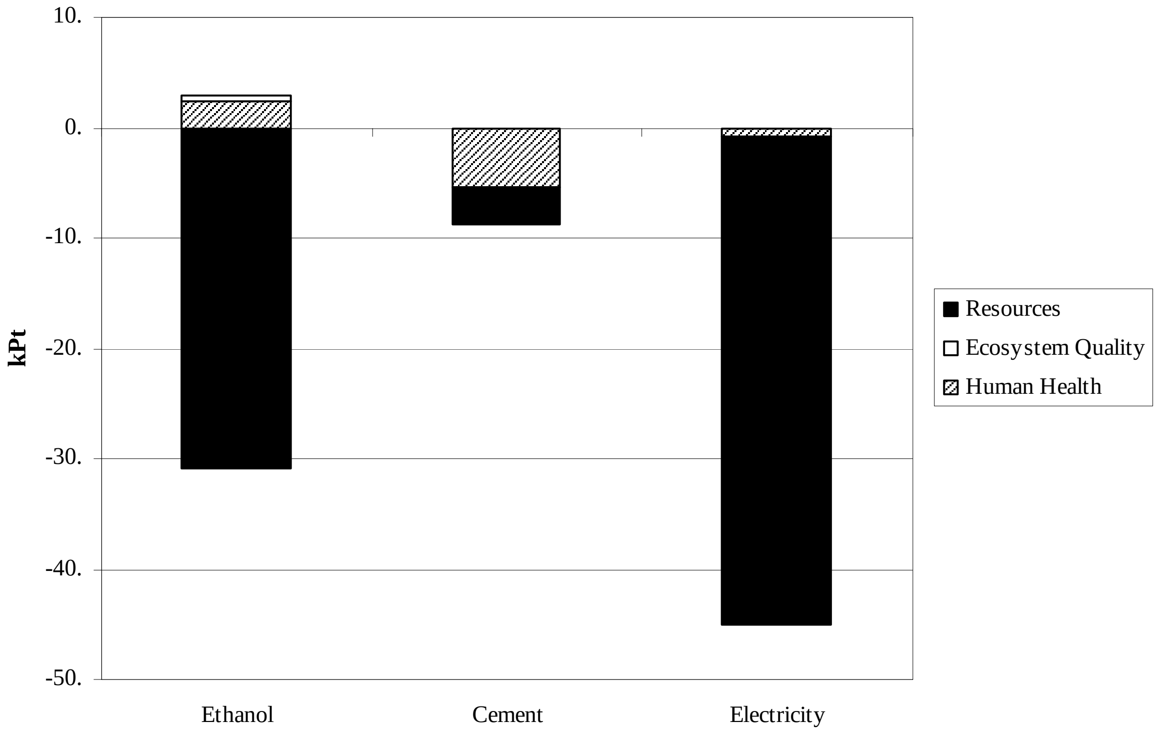

Fig. 6: A comparison of weighted impacts reduced by consuming 1000 tonnes of rice husks in different use systems

With regard to data issues, as a simplified LCA, data used in this study are not as high quality as the data used in the detailed LCA. As discussed in the inventory analysis section, the data were adapted to the Thai conditions where possible. The data used were also adapted to be as consistent with the goal and scope of the study as possible. More precision would lead to the more accuracy in LCI models in later stage; however, this is not the aim of the study.

Conclusion

Based on goal and scope defined and data available for this study, it can be concluded that using rice husks in the three systems investigated, i.e. electricity production, cement manufacture and cellulosic ethanol production, cause less environmental impacts on fossil fuels consumption and climate change compared with the conventional systems such as the Thai grid production, ordinary Portland cement and petrol production. However, they cause higher environmental impacts on some other indicators analyzed.

For the electricity generation system, the most environmentally preferable disposal option of rice husk ash produced from the rice husk power plants suggested is the use of the ash in light weight concrete block production. It is also suggested that rice husk ash from the rice husk power plants should never be disposed of in landfill because there is no environmental credit gained by disposing of the ash in this way. The ash from the rice husk power plants should be sent to other ash users, in this way it can also give added values and environmental credit to the rice husk power plants.

In a comparison of all rice husks use systems, the most environmentally favourable rice husks use system in fossil fuels consumption is the use in power plant compared to the use in cement manufacture and cellulosic ethanol production. This is because using rice husks in electricity production has the highest efficiency in the substitution of fossil fuels. In electricity production, rice husks can be used as a fuel to replace fossil fuels totally. While in cement manufacture the husks are used to substitute only 20 percent of coal used in the process. Though cellulosic ethanol can be used as an alternative fuel to substitute for petrol, their production processes are very different. In the production process of cellulosic ethanol, the rice husks need to be pre-treated before being distilled and the pre-treatment process requires various inputs and high energy consumption. Moreover, the ethanol distillery generates solid residues which then get burned to produce heat and electricity. This makes the cellulosic ethanol option less efficient in the substitution of fossil fuels.

In climate change, the best use of rice husks is the use in cement manufacture compared with the other uses. This is because in cement manufacture, rice husks are used to replace coal and that helps to reduce CO_2 emitted from burning coal. In the electricity system, the environmental impacts of rice husk power plants are compared with the impacts of the production of the Thai grid which has a large share of power generation from natural gas (approximately 66 %) (Amornkosit 2007). Natural gas is considered clean in terms of green house gases contribution. Therefore, with respect to climate change, using rice husks in cement manufacture is preferred to using them in power generation. In the cellulosic ethanol system, producing ethanol from rice husks seems to provide very little help in reducing green house gases compared with the petrol production. However, this work does not analyze the impacts caused by using cellulosic ethanol produced from rice husks in vehicles compared with the use of petrol and this will be investigated in a future study.

Acknowledgements

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Consequences of increased biodiesel production in Switzerland: Consequential Life Cycle Assessment (CLCA)

J. Reinhard[1], R. Zah[1]
[1]EMPA, Dübendorf, CH-8600 Dübendorf, Switzerland, juergen.reinhard@empa.ch

Keywords: Consequential LCA, system expansion, biodiesel

Abstract

This study analyses the direct and especially the indirect environmental impacts to be expected if Switzerland should increasingly produce biodiesel (RME) domestically. In order to take into account possible future consequences, what-if scenarios have been developed in co-operation with the Federal Office of Agriculture (FOAG) and assessed by means of a consequential LCA. This approach uses system expansion to include the consequences of a decision, thus avoiding allocation of co-products. This implies that the inputs and outputs are entirely attributed to biodiesel production and the product system is subsequently expanded to include the marginal products affected.

In summary, the overall environmental impacts of an increased RME production in Switzerland rather depends on the environmental scores of the marginal replacement products on the world market, than on local production factors. It is therefore crucial to consider at whose expense an increase in biodiesel production can be achieved, e.g. expansion into natural areas, displacement of other crops or the increased energetic utilization of the available edible oil, and what co-products are caused in addition. If, for example, barley instead of wheat is displaced by increased rape cultivation in Switzerland, the environmental scores of RME production decrease. Otherwise, if the possible marginal product on the world market for protein meal is switched from soybean meal Brazil to soybean meal USA, the environmental impacts of all analyzed scenarios would increase remarkable.

Introduction

Today, transportation accounts for 30% of the world's fossil fuel consumption and causes about 23% of total GHG emissions (Robert 2007). This leads governments to consider the use of alternative fuels in the transport sector to reduce GHG emissions. Fuels derived from biomass, so-called biofuels, are not only renewable but seem also to represent a promising alternative to fossil fuels on the short term. Biofuels are made from plant matter and other renewable feed stocks and are sufficiently similar to fossil fuels to provide direct substitution (Jungbluth, Chudacoff *et al*. 2007). The most widely used transport biofuels are ethanol and methyl ester (XME), which is also known as biodiesel.

The direct environmental impacts of biodiesel have been investigated extensively in various attributional Life Cycle Assessment (LCA) studies (Holden and Hoyer 2005; Ramesohl, Arnold *et al*. 2006; Schindler and Weindorf 2006; Zah, Böni *et al*. 2007) on a local and on a global scale. However, little knowledge exists with respect to the indirect local and global consequences. The production of biodiesel is strongly intertwined with other uses of land like nature conservation (Wiesenthal, 2006), supply of food (van den Broek, Treffers *et al*. 2002) and the production of biomaterials (Dornburg, Lewandowski *et al*. 2004). Moreover, the increased production of biodiesel causes additional co-products such like oil meals and glycerine, which affect the production of alternative products on the world market. For a sound assessment of the total environmental impacts of producing biodiesel, it is therefore necessary to address also indirect impacts, which take place outside biodiesel´s value chain.

Method / Approach

LCA is a method for analyzing and assessing environmental impacts of a material, product or service along its entire life cycle (ISO 2006). Two main approaches are distinguished: the attributional and the consequential approach.

Attributional LCA (ALCA) is defined by its focus on describing the environmentally relevant physical flows to and from a life cycle and its subsystems (Ekvall and Weidema 2004). Within an ALCA, the system investigated is limited to a single full life cycle from cradle to grave. Hence, co-production has to be treated by applying allocation factors. Furthermore, the attributional approach uses average data in order to attribute the average environmental burdens for producing a unit of the product in the system (Ekvall and Weidema 2004).

Consequential LCA (CLCA) is defined by its aim to describe how environmental impacts will change in response to possible decisions (Ekvall and Weidema 2004). In contrast to ALCA, the system within a CLCA is not limited to a specific life cycle. Instead of allocation, the consequential approach uses system expansion to include additional life cycles and products affected by a change of physical flows in the respective life cycle. Marginal data instead of average data is used for the consequential approach. Marginal data stays for the product, resource, supplier or technology, which is most sensitive to changes in demand.

Scenarios analyzed

According to the FOAG, the increased production of RME in Switzerland would occur at the expense of i) other crops and ii) the available edible rape oil. Both cases induce further consequences since it is assumed that the demand for a displaced crop or product will be compensated for by increasing imports of an equivalent crop or product from foreign countries. In accordance to the FOAG, the following scenarios are analysed (Tab. 1).

In order to evaluate the burdens related to the displacement of a specific crop, each branch of consequence is analysed down-the-line by means of the determined functional unit (38,3 GJ energy at regional storage in Switzerland)[11]. Since price elasticity is not taken into account it is assumed that the equal amount displaced will be compensated for. With regard to pasture and meadow, potatoes and feed grain, it is assumed that the identical crop displaced will be compensated for. The displaced amount of edible oil, in turn, is expected to be supplied by imports of (i) rape oil or (ii) sunflower oil from Europe or (iii) palm oil from Malaysia.[12]

Scenario-related system delimitation

(0) Diesel

In this reference scenario it is assumed that no additional imports of biodiesel take place. The full Swiss demand is fulfilled with imported low-sulphur diesel.

(1) Domestic RME production

Within the attributional scenario the system is strictly limited to the defined life cycle. Consequently co-products are handled by allocation. The no-allocation scenarios, in turn, include the co-products, i.e. ascribe the environmental impacts of the co-products fully to the determining product. The consequential systems are further enlarged to the consequences induced by co-products, i.e. glycerine and oil meal and the consequences on the agricultural stage. However, the required system delimitation changes with respect to the assumption how the increased demand for the required vegetable oil is met, i.e. displacement of other crops or increased utilization of the available vegetable oil.

[11] The functional unit refers to the net calorific value of RME, which could be produced from one hectare land in Switzerland.

[12] It is worth noting, that the fatty acid composition of rape seed, sunflower, soybean and palm oils are not the same. However, according to Schmidt & Weidema (2008) they are substitutable within the most important applications (frying oil/fat, margarine, shortening and possibly salad oils) and hence, they are treated here as equivalent.

Tab. 1: Scenarios analysed (source: according to FOAG).

Scenario	**System delimitation**	**Increased RME production in CH is met**	**Consequence**	**Compensation**	**in country**	**by**	**Scenario-Label**
(0): Diesel is imported	attributional	-	-	-	-	-	REF
(1): Domestic RME production	attributional	-	-	-	-	-	RME_ATT
	no allocation	-	-	-	-	-	RME_NO
	consequential	(1.1) at the expense of other crops (displacement)	less potatoes	import potatoes	Israel (ISR)	expansion	RME_POT_IS
					Europe (RER)	expansion	RME_POT _RER
			less barley	import barley	Europe (RER)	expansion	RME_BAR _RER
						intensification	RME_BAR _RER_INT
					Canada (CAN)	expansion	RME_BAR _CAN
			less wheat	import wheat	Europe (RER)	expansion	RME_WHE _RER
						intensification	RME_WHE _RER_INT
					Canada (CAN)	expansion	RME_WHE _CAN
			less grain maize	import grain maize	Europe (RER)	expansion	RME_MAI _RER
					USA (US)	expansion	RME_MAI_US
		(1.2) at the expense of the available rape oil	less rape oil	import rape oil	Europe (RER)	expansion	RME_OIL _RAPE
				import sunflower oil	Europe (RER)	expansion	RME_OIL _SUN
				import palm oil	Malaysia (MY)	expansion	RME_OIL _PALM

(1.1) Displacement

An increased cultivation and extraction of oil crops cause a corresponding growth in the production of oil meal and glycerine. According to the FOAG, the additional glycerine is exported to Europe, where it is assumed to reduce the industrial production of glycerine from epychlorhydrine. The system delimitation induced by oil meals has previously been dealt with and described by Weidema (2003), Dalgaard (2007) and (Schmidt and Weidema 2008). The FOAG but also Weidema (2003) and (Schmidt 2008b) determined soybean meal from Brazil as the protein source most sensitive to changes in demand. However, when soybean meal is displaced, the output of the dependant co-product soybean oil is also affected. According to Schmidt and Weidema (2008), market response to that will most likely be an increase in production of the marginal vegetable oil, i.e. rape oil (Fig. 1).[13] The increased production of rape meal lead to an additional amount of oil meal and again the production of soybean meal in Brazil is affected.

The reduction in soybean meal is calculated by means of the difference in the protein content between the co-produced meal and soybean meal. The sole application of the protein content is a simplification of the reality, since not merely the protein content, but also other influence factors such as fatty acid compositions and the energy contents determine the application of a specific meal (Schmidt 2008b).

[13] Schmidt and Weidema (2008) determined palm oil as the marginal oil on the global market. However, the prices for palm oil have increased rapidly within the last year (Bradsher 2008). Thus, we assumed rape oil to be the marginal oil.

However, according to the FOAG, no general fodder unit is defined for Switzerland and since each animal transforms a different part of the energy only the protein content was taken into account.

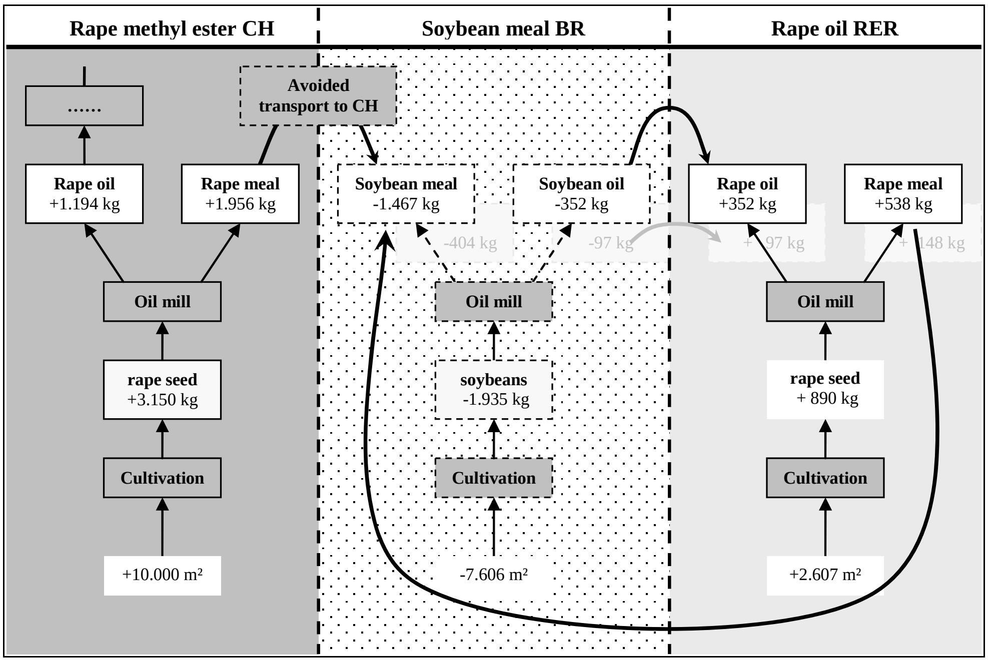

Fig. 1: Soybean meal-rape oil loop caused by the additional production of 38.3 GJ RME in Switzerland at the expense of other crops. The shaded boxes represent the start of the second loop (source: Reinhard 2008).

(1.2) Increased utilization of the available rape oil

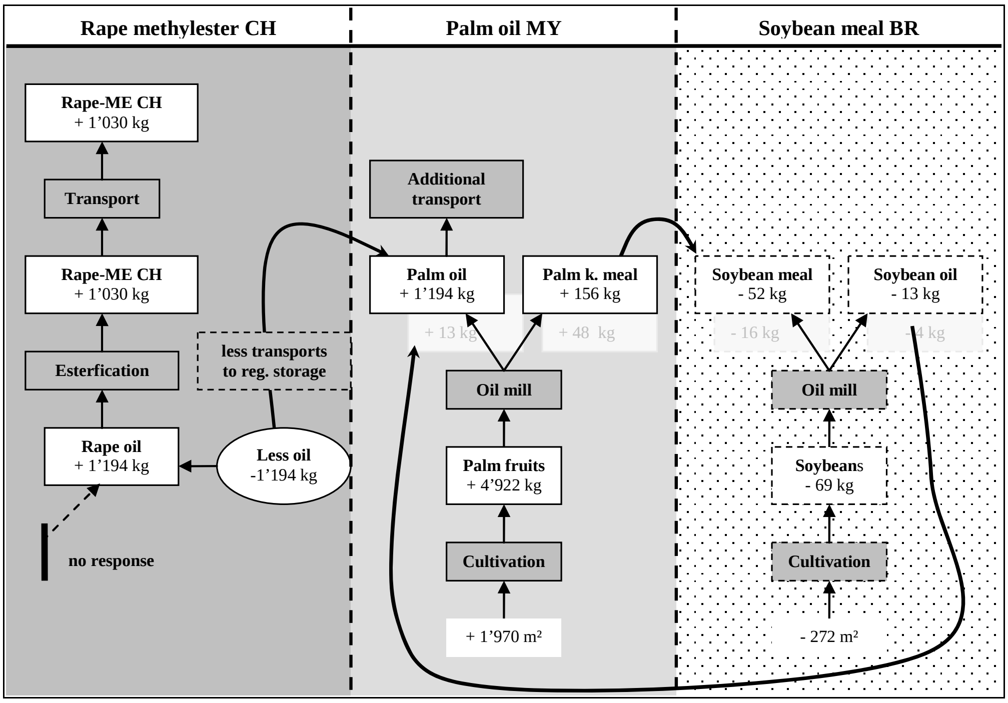

Fig. 2: Palm oil-soybean meal loop caused by the additional production of 38.3 GJ Rape-ME CH. The shaded boxes represent the start of the second loop (source: Reinhard 2008).

If the increased production of RME occurs at the expense of the available oil, the cultivation of oil crops and the extraction to oil is not affected and is hence excluded from the system boundaries. Since the extraction of oil is not affected, no additional amount of oil meal is produced. Instead, less edible oil is available for consumption. According to the FOAG, this would increase the production of palm, rape or sunflower oil. However, the increased production of those oils will cause an additional amount of oil meal. Taking account of the respective protein content the additional meal is assumed to reduce the production of soybean meal in Brazil. Fig. 2 shows exemplary the system delimitation induced, if the increased production of RME in Switzerland occurs at the expense of the available rape oil and the corresponding lack in rape oil is compensated for by increased imports of palm oil from Malaysia.

System boundaries on the agricultural stage

Corresponding effects to an increased demand for a specific crop are displacement, intensification and expansion (Kløverpris and Wenzel 2007).

Displacement substitutes one crop with another and is primarily assumed to occur in countries which face physical and regulatory constraints. In this study, merely the first displacement step in Switzerland is modelled. Further displacement steps in foreign countries are not assessed. The rationale reason is that the related replacement mechanism is simply too complicated to be modelled down-the-line. Thus, primarily expansion and if adequate data is available, intensification, are assumed to be the possible system reactions. This is regarded as a good proxy for the actual effects that are taking place since the factor of crop displacements will decrease for each further replacement step.

Intensification increases the yield of a given area by additional inputs, i.e. optimization of production and technological development (Kløverpris and Wenzel 2007). In this study, intensification is modelled by calculating the difference between extensive and intensive production on the basis of Swiss LCI data from the ecoinvent database. Thus, intensification is not merely driven by applying an additional amount of fertilizer but by the whole difference in the cultivation practice. However, this approach indicates a linear increase in yields, taking not into account that the increase in yield diminish with increased inputs (Fig. 3). In this context it also is to mention, that the possible increase in yield is determined by the yield before cultivation is intensified. Nevertheless, the results are expected to be valid as long as (i) the geographical conditions are comparable and (ii) country specific crop yields do not differ on a large scale.

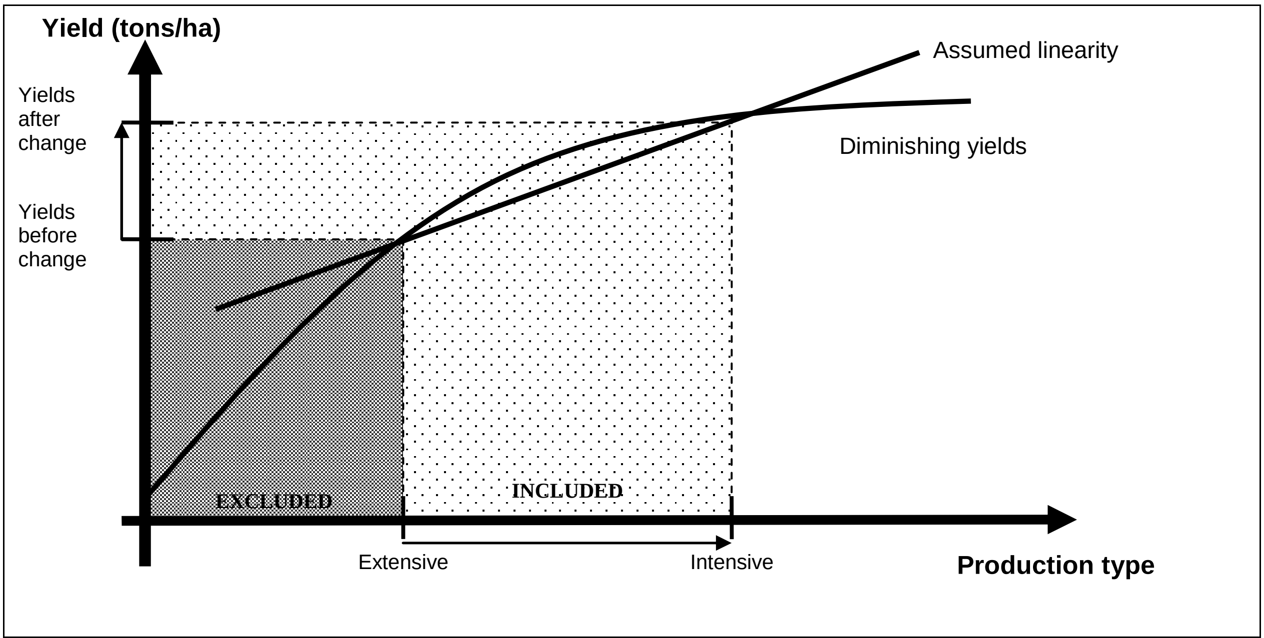

Fig. 3: Derivation of the LCIs to model intensification for a specific crop (source: Reinhard 2008).

Expansion is defined by the transformation of a specific land type, e.g. natural areas or fallow land, into arable land. If an increased demand for a specific crop is met by the transformation of natural areas into arable land, the system must be enlarged to include (i) the avoided interventions inherent to the alternative land use, i.e. commonly land under natural vegetation and (ii) the emissions related to the transformation (Schmidt 2008a). In this paper (i) is not included since sensitivity analysis proved their influence as insignificant. With respect to (ii), those emissions are not directly included in the analysis. However, using data from (Schmidt 2008b) a sensitivity analysis is applied in the discussion section in order to evaluate the importance of the emissions from land use change.

Impact assessment

In order to model the required product systems, Life Cycle Inventories (LCIs) from the ecoinvent database were used (Frischknecht, Althaus *et al*. 2007). The environmental impacts were assessed by means of characterized CML indicators (Guinée 2001), land (Schmidt 2008b)occupation and the Swiss method of ecological scarcity (Frischknecht, Steiner *et al*. 2008). In this paper, merely GHG emissions and aggregated environmental impacts (UBP 06) are shown.

Results

The study shows different trends in environmental impacts, depending on the assumption how the increased demand for the required rape oil is met: (i) displacment of other crops or (ii) increased utilization of the available rape oil.

(1.1) Displacement

When the increased production of RME in Switzerland is realized at the expense of other crops, the effective environmental impacts are determined by (i) the initial impacts caused by the value chain of RME production in Switzerland and the related co-products, (ii) the consequences caused by the additional co-products, i.e. rape meal and glycerine and (iii) the difference between the displaced domestic and the additional foreign crop production.

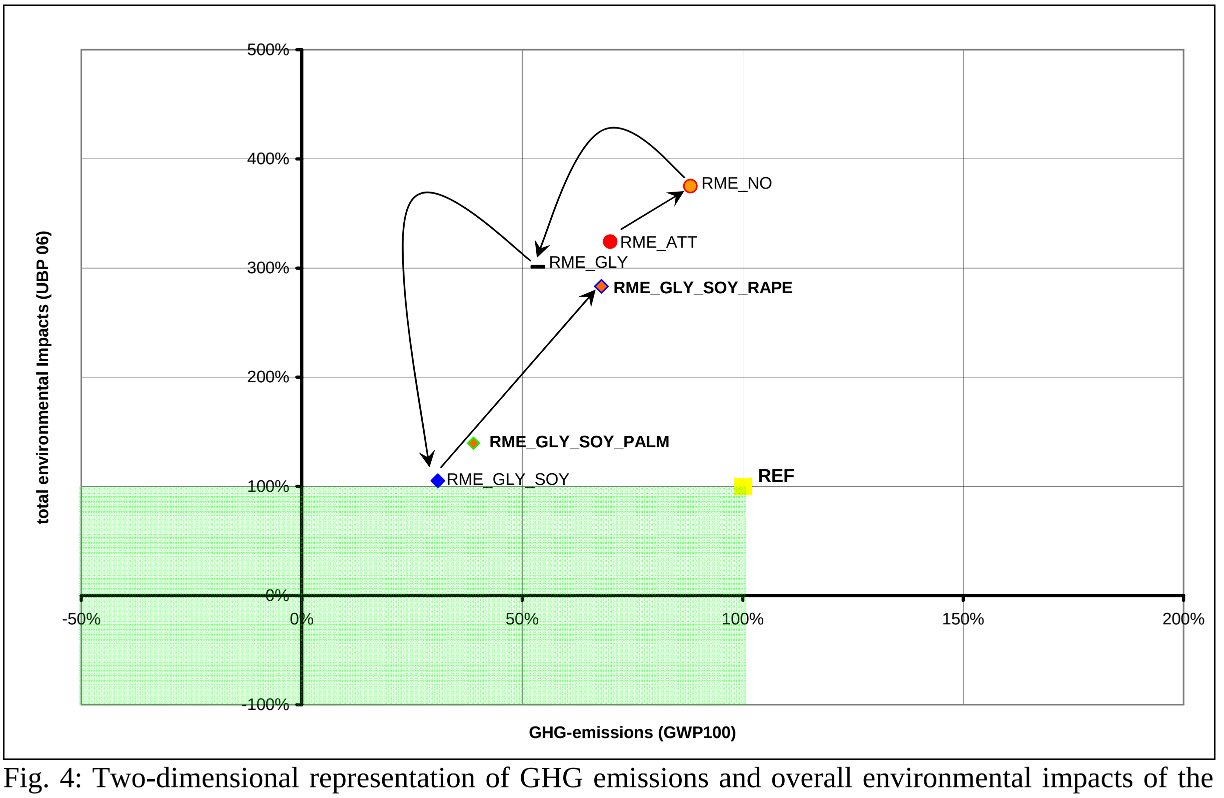

Fig. 4: Two-dimensional representation of GHG emissions and overall environmental impacts of the attributional scenario (RME_ATT) and the consequences induced by co-products. Values are relative to the fossil reference. The black arrows shows the development of the environmental impacts if the studie system is gradual enlarged to (i) the co-products (RME_NO), (ii) the avoided glycerine production (RME_GLY), (iii) the avoided soybean meal production (RME_GLY_SOY) and (iiii) the additional rape oil production (RME_GLY_SOY_RAPE). Scenarios in the green area show a better environmental performance than the fossil reference.

With respect to (i), RME production in Switzerland causes less GHG-emission than the fossil reference but contributes more impacts with respect to the overall environmental evaluation (Fig. 4).

Regarding (ii), the crediting effects related to the avoided production of glycerine in Europe lead to a reduction in GHG emissions and UBP. The avoided production of soybean meal in BR further contribute negative to both GHG emissions and in particular UBP and decreases several other environmental impact factors. However, those crediting effects are diminished due to the related growth in the production of rape oil. If palm oil is assumed to be the marginal oil, the environmental impacts are lower (RME_GLY_SOY_PALM). The reason is that less palm oil is produced in addition and scale effects reduce the impacts per kg palm oil produced. All in all, the outcomes are dominated by the impacts of (i) RME production in Switzerland (primarily rape cultivation) and (ii) the additional production of the marginal oil.

Regarding (iii), if in addition the displacement on the agricultural stage are mentioned, the results show a broad distribution (Fig. 5). The difference of a respective scenario to the red lines is the environmental difference between domestic and foreig crop production. With respect to expansion, most of the analyzed scenarios show that the additional agricultural production in foreign countries contributes more impacts than the domestic cultivation of the crop displaced. The reasons for this are partially low crop yields (e.g. wheat and barley from Canada) and partially the intensive fertilizer use in foreign countries (e.g. wheat and barley from Europe). The compensation of the increased agricultural production by intensification leads to lower environmental impacts than expansion of the agricultural area. This might be explained by the fact, that only the additional environmental impacts caused by the intensification have been accounted for. It should be noted, that the potential for large-scale and rapid intensification is much more limited than for expansion. Finally, it would probably be a mixture between expansion and intensification, which is used to compensate the increased demand for a specific crop.

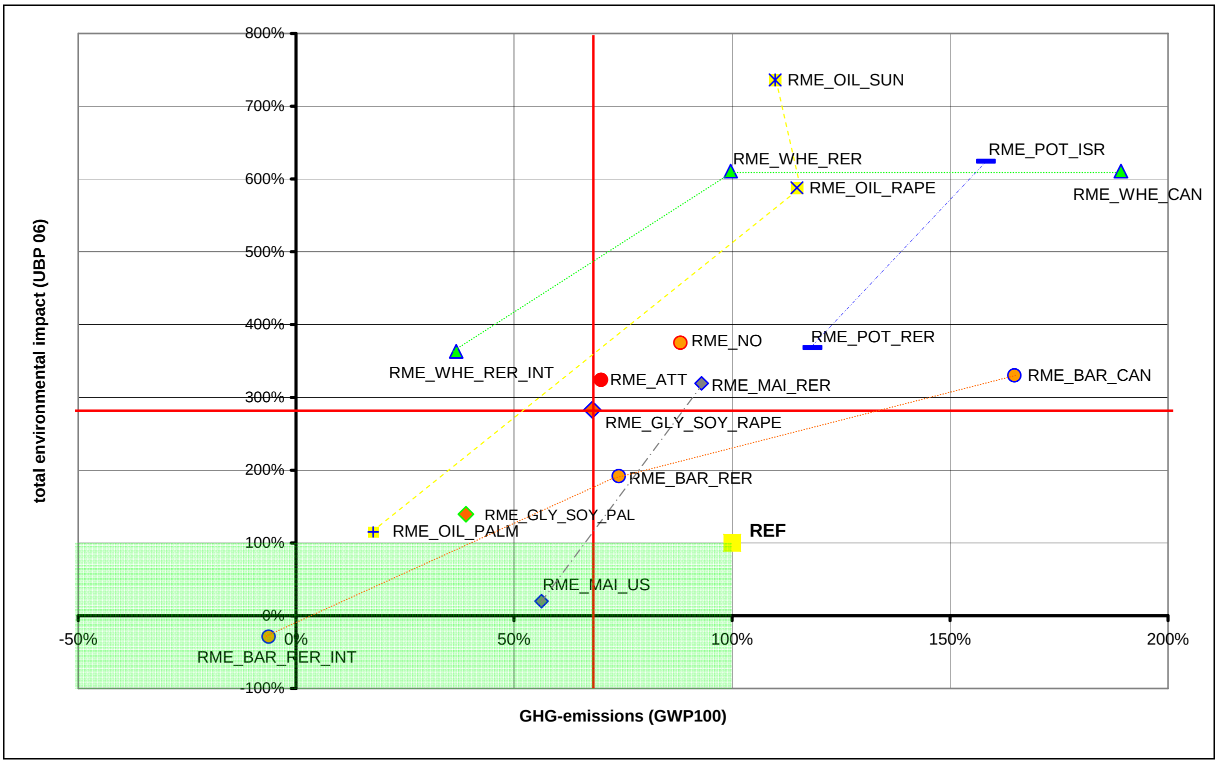

Fig. 5: Two-dimensional representation of GHG emissions and overall environmental impacts of all scenarios analysed. Values are relative to the fossil reference. Scenarios in the green area have a better environmental performance as regards both GHG emissions and the overall environmental evaluation.

The additional transport does seldom cause more than 10% of the impacts related to the cultivation even if the product displaced is imported from Canada. The reason is that the transport by transoceanic tanker is much more environmentally friendly than the transport by truck. However, if crops with a high mass would be displaced in Switzerland, transport would become more important. In this study, for instance, it is primarily the RME production increased at the expense of potatoes or grass which causes significant impacts due to increased imports from foreign countries.

(1.2) Increased utilization of the available rape oil

The increased production of RME at the expense of the available rape oil causes a shift off the central life cycle up to another life cycle. Finally, the resulting impacts are determined by the vegetable oil, which compensates for the lack of rape oil in Switzerland (yellow squares in Fig. 5). The additional production of sunflower and rape oil contribute significantly to the overall environmental evaluation and increase also GHG emissions. Even though this impact is diminished due to the corresponding decrease in soybean meal production, most environmental impact factors and also the overall environmental evaluation display significant higher impacts than the production and use of the fossil reference. The additional production of palm oil cause fewer impacts than rape and sunflower oil with respect to the most environmental impacts factors and also regarding the aggregated assessment. Somehow or other, under the current circumstances the production of RME at the expense of the available rape oil contribute more to the overall environmental evaluation than the production, import and use of the fossil reference.

Discussion

All in all, most of the scenarios analysed show higher impacts than the production and use of fossil diesel with respect to GHG emissions, mid-point environmental indicators and aggregated environmental indicators.

However, the emissions resulting from land use changes (LUC) have not been taken into account so far. Tab. 2 shows (i) the assumed LUC in the countries affected, (ii) the GHG emissions caused by a specific LUC attributed to 20 years[14] and (iii) the percentage change per scenario using the fossil reference as a baseline.

Tab. 2: Influence of LUC on the results for GHG emissions. The upper part of the table shows the GHG emissions caused by a specific LUC attributed to 20 years (source: Schmidt 2008b). The lower part gives the percentage change of a specific scenarios on the x-axis in Fig. 5 (GHG emissions), which is caused by a specific LUC.

Region	Europe		Malaysia		Brazil		Canada/USA /Israel	Sum
Transformation from	100% Set-aside	100% Grassland	50% Sec. forest	50% Grassland	95% Savannah	5% Sec. forest	100% Grassland	-
Transformation to	Rape seed or Sunflow.	Feed grain	Oil palm	Oil palm	Soybean	Soybean	Feed grain or Potatoes	-
GWP (100) [kg CO_2 eq./ha y^{-1}]	4'750	4'500	21'850	1'650	15'150	39'800	4'500	-
Scenario	Percentage change of the scenarios on the x-axis in fig. 5							
RME_POT_RER	54%	120%	0%	0%	-443%	-61%	0%	**-329%**
RME_POT_ISR	54%	0%	0%	0%	-443%	-61%	154%	**-296%**
RME_BAR_RER	54%	113%	0%	0%	-443%	-61%	0%	**-336%**
RME_BAR_RER _INT	54%	0%	0%	0%	-443%	-61%	0%	**-450%**
RME_BAR_CAN	54%	0%	0%	0%	-443%	-61%	288%	**-162%**
RME_WHE_RER	54%	106%	0%	0%	-443%	-61%	0%	**-344%**
RME_WHE_RER _INT	54%	0%	0%	0%	-443%	-61%	0%	**-450%**
RME_WHE_CAN	54%	0%	0%	0%	-443%	-61%	330%	**-120%**
RME_MAI_RER	54%	154%	0%	0%	-443%	-61%	0%	**-295%**
RME_MAI_US	54%	0%	0%	0%	-443%	-61%	128%	**-321%**

[14] The time horizon was chosen in accordance with the IPCC Guidelines (2006).

Region	Europe		Malaysia		Brazil		Canada/USA /Israel	Sum
RME_OIL_RAPE	163%	0%	0%	0%	-443%	-58%	0%	**-337%**
RME_OIL_SUN	253%	0%	0%	0%	-384%	-54%	0%	**-185%**
RME_OIL_PALM	0%	0%	64%	-5%	-15%	-2%	0%	**41%**

It appears that the benefits from avoided land transformation in Brazil dominate the outcomes. For example, when all land use changes are considered for RME_BAR_CAN, the GHG emissions for the scenario would decrease by -162% to approx. 0% in relation to the fossil reference. The reasons for this are (i) the substitution ratio between rape and soybean meal, (ii) the low soybean yield and (iii) the high GHG benefit caused by the preservation of carbon rich rain forest in Brazil. Regarding (i), as stated prior the sole application of the protein content is a simplification. Thus, the amount of soybean meal substituted is possibly overestimated meaning that the GHG-benefit is possibly lower than calculated.[15] With respect to (iii), in a global perspective producing biofuels at the expense of low carbon land and simultaneously avoid the devastation of carbon rich rain forest appears strictly limited.

All in all, the outcomes strongly depend on the applied system expansions, i.e. the marginal meal and oil taken into account, and the related land transformations. For example, if the marginal oil would switch from rape to palm oil, the emissions for land use change would increase strongly. The reason is that the benefits resulting from avoided soybean cultivation in Brazil are compensated by the corresponding emissions from expansion of oil palm cultivation. In this context, the emissions from land use change appear very important. Thus, a clear determined methodology for their inclusion is urgently needed. One main source of uncertainty is the difficulty of discounting the emissions from soil organic carbon resulting from land transformation on a definite time scale. Possible time scales are for example, the cultivation time of a certain crop or the time period until a new equilibrium in soil carbon occurs.

The study shows that the approach to system delimitation matters. Attributional LCA accounts for the environmental impacts of the central life cycle and thus lacks possible consequences resulting from an increased use of the product under study. CLCA, in turn, provides information of the consequences follow-on a decision and goes thus far beyond an attributional perspective. However, the results of a CLCA strongly depend on applied system expansions. Hence, the arbitrariness related to allocation within the attributional methodology is not avoided but rather shifted to the identification of (i) the marginal products on the world market, (ii) the relevant parameters to calculate the substitution and (iii) the possible feed back mechanisms.

Conclusion

In sum the environmental impacts of an increased biodiesel production in Switzerland rather depends on the environmental scores of the marginal replacement products on the world market, than on local production factors. Thus, it is not only the manner in which biodiesel is produced. In fact one also has to consider at whose expense an increase in biodiesel production can be achieved, e.g. expansion into natural areas, displacement of other crops or the increased energetic utilization of the available edible oil, and what co-products are caused in addition.

In general, most of the analysed scenarios show higher environmental impacts than the fossil reference as regards both GHG emissions and the overall environmental evaluation. In this context, the main environmental impacts are caused by agricultural cultivation, i.e. both within the central life cycle and in the life cycles to which the system is enlarged, whereas transport and conversion, in turn, seldom cause more than 10% of the impacts related to the cultivation. However, if the additional/avoided emissions from land transformations are taken into account GHG emission would decrease. In this

[15] In addition to the protein content, Schmidt (2008b) used Scandinavian Feed Units and the energy content to calculate the substitution between rape and soybean meal. This result in an overall substitution ratio of 0.76 compared to our ratio of approx. 1 (both including the feed back loop).

perspective, increased RME production in Switzerland avoids the transformation of carbon rich rain forest and savannah in Brazil and thus causes a potential GHG benefit.

The potential of domestic biofuels is limited today and will remain so in future. On a global scale, increased production of biofuels would influence the food self-sufficiency a country and would decrease natural habitats. From a long-term environmental perspective it would therefore seem wise, to focus the production of biofuels on feedstock decoupled from the global food and feed markets. Examples are biogenic waste or non-edible energy crops that grow specifically on degraded land.

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Effect of Canadian bioenergy production from agriculture on life-cycle greenhouse gas emissions and energy

Brian G. McConkey[1], Stephen Smith[2], Ravinderpal Gil[2], Suren Kulshreshtha[3], Cecil Nagy[3], Murray Bentham[4], Darrel Cerkowniak[4], Bob MacGregor[2], Marie Boehm[4]
[1,2,4]Agriculture and Agri-Food Canada, [1]Swift Current, SK, [2]Ottawa, ON, [4]Saskatoon, SK [2]University of Saskatchewan, Saskatoon, SK, Canada, mcconkeyb@agr.gc.ca

Keywords: LCA, Canada, bioenergy, biofuel, energy, greenhouse gas, agriculture, crops

Abstract

For greenhouse gas (GHG) mitigation and rural development reasons, Canada has policies in place to increase biofuel production using feedstocks from agriculture. To answer the question if bioenergy policies are sensible from an environmental perspective, life-cycle analysis (LCA) was used to determine the GHG and energy impact of an aggressive bioenergy policy for 2017. The aggressive bioenergy policy did not affect overall energy required by Canadian agriculture but did provide GHG reduction benefit of about 24 million tonnes of carbon dioxide (CO_2) equivalent – primarily because of the fossil fuel replacement with bioenergy. Aggressive bioenergy production decreased Canadian food exports modestly. The effect of bioenergy production was mitigated by the fact that straw was the most economical bioenergy feedstock. Distiller dried grain and canola meal co-products from biofuel production from grain became important livestock feeds. This also helped reduce the effect of bioenergy production on food. The effect of aggressive bioenergy production policy on GHG emissions from potentially increased deforestation in Canada and from GHG emission and other environmental impacts of near total removal of cereal straw were not considered fully but these could easily negate any environmental benefit of bioenergy production.

Introduction

Among global drivers acting on society are concerns about rising price of petroleum, the security of long-term supply of petroleum, low returns for primary agricultural production, and dangerous climate change due to increasing greenhouse gases emissions to atmosphere. In reaction to these drivers, there have been numerous policies implemented in Canada and elsewhere in the world to use feedstocks from agriculture as biogenic energy sources to replace fossil fuel use. Unlike many other developed countries, Canada is a net exporter of energy (as petroleum, coal, natural gas, electricity, wood pellets), therefore energy security is not an important immediate concern for Canada. The life-cycle energy and GHG benefit of biogenic energy production from agricultural feedstocks, then, is critical to assess the value of bioenergy policies for Canada. Using agricultural land to produce feedstocks for bioenergy instead of traditional crops will affect other parts of the agricultural system. For example, use of grains to produce ethanol will increase the cost of feed grains for livestock and thereby affect the amount of livestock feeding. This, in turn, will affect the energy used for and GHG emissions from livestock production. Therefore, it is necessary to include the effects on the entire inter-related agricultural system within the LCA.

Method / Approach

The project involves linking the economic model of Canadian agriculture linked to a general energy budgeting and greenhouse gas accounting models (Fig 1). The Canadian Regional Agricultural Model (CRAM) is used for this analysis. CRAM is a sector equilibrium model for Canadian agriculture which is disaggregated across both commodities and space (Horner at al., 1992). CRAM is a non-linear optimization model maximizing agricultural producer plus food consumer surplus. The basic commodity coverage is grains and oilseeds, forage, beef, hogs, dairy and poultry (horticulture is excluded). The Canadian Economic and Emissions Model for Agriculture (CEEMA) is designed to estimate energy budget and GHG emissions from CRAM output of amount of included agricultural activities (Kulshreshtha *et al*. 2000; Kulshreshtha and Sobool 2007). Greenhouse gas emissions from livestock and soils are derived from methods used for the Canadian National GHG Inventory (Environment Canada 2006) while those from farm energy use and embodied in inputs (machinery, pesticides, fertilizers, etc.) are based on LCA of set hypothetical farms having regionally representative areas and machinery complements (Dyer and Desjardins 2003, 2007). The carbon (C) change on agricultural land can represent an important source or sink of atmospheric carbon dioxide and needs to be included in the GHG budget. The current land C change is complex because it is the cumulative effect of land use or land management changes over the past several decades as well as current land use and management. It can be considered both a direct and indirect emission – indirect for causes of C change from the past unrelated to current management and direct for C change from current practices. Carbon change was estimated using national inventory methods (Environment Canada 2006).

Maize is an important feed grain for livestock and a feedstock for ethanol production. Canada is currently a net importer of maize, almost entirely from the US. This production lies outside the system boundaries used for the GHG analysis. Therefore, the GHG reductions from substitution of fossil fuels with ethanol from imported maize are an overestimate of real GHG reductions because the GHG emissions for production of the imported maize were not included. Nevertheless, these biased GHG emission reductions are important for Canadian policy development as they correspond to GHG reporting under the international climate change treaties, including the Kyoto Protocol.

A number of future bioenergy scenarios for 2017 were considered based on assumed price of carbon and oil. In this paper we will only discuss a relatively aggressive policy for biogenic energy production with 20% of gasoline replaced with ethanol (8.8 billion L), 8% of diesel replaced with biodiesel (1.44 billion L), and 20% of coal used for generating electricity replaced with biomass (33.4 billion kWh). Collectively these represent 33.5 PJ of energy provided from agricultural feedstocks. The basis for this scenario is an assumed high oil price ($120/barrel) (all values in Canadian $, CAN$1 ≈ US$0.9) that produces demand for ethanol and biodiesel production and moderate carbon price ($20/tonne CO_2) that produces demand for reduced coal usage. These energy needs were assumed to be mandated by regulation so that the production had to be met. Ethanol could be produced from either lignocellulosic feedstocks (grass or coppiced hybrid poplar or willow) or from grain (maize or wheat). This scenario was compared with existing medium-term baseline for agriculture that does not have significant bioenergy production (Agriculture and Agri-Food Canada 2008). This outlook includes bioenergy production to meet the current Canadian mandate for ethanol (5% of gasoline) and biodiesel (2% of diesel fuel) (totalling 5.9 PJ of energy).

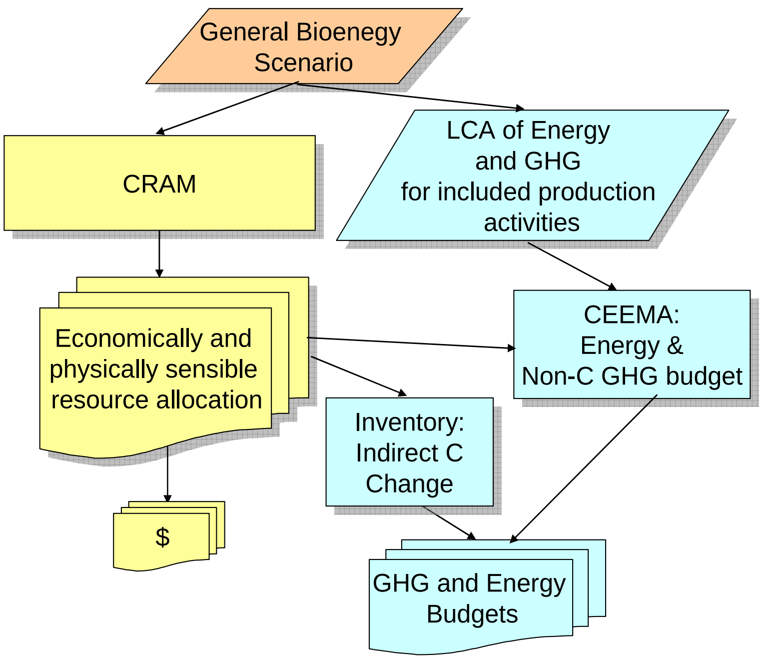

Fig. 1. Outline of project

Results and Discussion

There was little difference in the energy required with and without enhanced bioenergy production from agricultural feedstocks (data not shown). However, there was almost 23.6 million tonne CO_2 equivalent (i.e. the global warming potential of each GHG converted to that of CO_2) reduction in GHG emissions for aggressive bioenergy production (Tab. 1). This occurred because there was increase in land carbon due to conversion of 0.6 million hectares from annual crops to perennial biomass crops (predominantly grasses), reduction in livestock production, and lower amount of food processing. There was however increase in GHG emissions for on-farm inputs, especially fertilizer.

To estimate the effect of maize imports, the GHG benefit of bioenergy substitution with ethanol from maize was excluded assuming all the imported maize was used to meet ethanol production targets (Tab. 1). This better approximates the life-cycle GHG benefit of bioenergy production for domestically produced feedstocks in Canada although it excludes a GHG benefit that would accrue to Canada under the boundaries relevant to reporting under the Kyoto Protocol.

Tab. 1: Greenhouse Gas Emission (millions of tonnes of CO_2 equivalent)

Source	Medium-term Outlook	Aggressive Bioenergy scenario
Land (carbon change and N_2O emissions)	33.7	31.5
Livestock production	31.3	30.2
On-Farm Energy	8.6	8.5
On-Farm Inputs	16.1	17.8
Off-Farm Transportation and Storage	1.0	0.9
Processing	35.0	29.5
Total Emissions	125.9	118.4
Reduction from bioenergy substitution for fossil fuels	-5.8	-21.9
Net total emissions relevant for Kyoto Protocol reporting	120.1	96.5
Reduction from bioenergy substitution by excluding imported maize used for ethanol production	0.0	-17.1
Approximate net total emissions without bioenergy substitution from imported maize	125.9	101.3

Canadian exports of grain and semi-processed grain products are decreased modestly with aggressive bioenergy production (Tab. 2). The increased export of distiller dried grain and solubles essentially substitutes for protein and food energy represented in decreased exports of legumes and oilseed meals. The export of cereals is essentially unaffected by bioenergy productions because the demand for straw maintains their production. The change in canola exports is well within typical interannual variability of Canadian canola exports and small within context of world oilseed trade so effects outside of Canada would lie well within normal market variation. Although not large relative to total global trade, the increased Canadian imports of maize could potentially marginally increase the global price of coarse grains. The expected market reaction would be reduced feeding of coarse grains to livestock that would generally decrease worldwide GHG emissions from agriculture.

Tab. 2: Effect of aggressive bioenergy production scenario on Canadian exports of selected grains and semi-processed grain products.

Grain	Exports (million tonnes)	Change from Medium-Term Outlook (%)
Wheat	17.6	-4
Canola	2.3	-19
Canola meal	3.6	-7
Pea	1.9	-23
Barley Malt	1.7	-38
Distiller dried grain and solubles	2.1	+1518
Maize	-5.8 (import)	+93%

Returns for crop production for aggressive bioenergy scenario are 20 to 100% higher than without this policy. The greatest improvement was in eastern Canada where maize is predominate cereal crop.

Food versus fuel?

Life-cycle analysis has shown that aggressive bioenergy production from agricultural feedstocks in Canada provides important energy and greenhouse gas benefit to Canada, the policy has merits from those perspectives. The main issue that society must confront is the impact on food resources. At the assumed $120/barrel oil, our study shows it is profitable to produce ethanol from grain without mandate or subsidies. Therefore, at high oil prices in an open market, food prices will have to rise to compete effectively with biofuel production. At lower oil prices, the lower cost lignocellulosic feedstocks are preferred over grain so less direct competition between biofuel and food although there remains indirect competition for agricultural land. Regardless of oil price, because of the relatively high value of oilseed feedstock for biodiesel, biodiesel production has no economic advantage and is produced at an economic loss to meet mandated biodiesel requirements. Economics strongly favoured biodiesel production from canola rather than from soybean. The feeding of livestock with distiller dried grain and canola meal co-products from biofuel production from grain helped reduce the effect of bioenergy production on livestock production.

Straw from cereals (maize, wheat, oat, barley) was the economically preferred lignocellulosic feedstock. The demand for biogenic energy consumed essentially all the available cereal straw not needed for livestock feed and bedding. The straw is used for both ethanol and to produce electricity. The former could be done profitably but the latter was not economic relative to burning coal. Therefore, only the mandated requirement for electricity was produced from biomass. About 1 million hectares of grass and only about 50 thousand hectares of coppiced wood added to the lignocellulosic feedstock supply. These were grown mainly to meet the shortfall in straw required for mandated electricity production from biomass. Since straw is a co-product of grain production, this reduces the effect of bioenergy production on grain exports compared to a situation where only dedicated biomass crops supply lignocellulosic feedstocks.

Effect of crop residue harvesting?

The potential loss of C from routine removal of straw was not considered in the inventory. There have been several relevant studies for Canada that show the losses of soil carbon of likely of about 0 to 0.3% of soil carbon per year for first 50 years after residue harvesting starts (Campbell *et al*. 1998; Ketcheson and Beauchamp 1978; Malhi and Lemke 2007). Given typically values for soil carbon in Canada, this translates to a large-area average of between 0 to 1000 kg CO_2 emissions from soil per year per hectare. Assuming a representative rate of 250 kg CO_2 per hectare over 16 M ha of cereal production, this would represent an emission of 4 million tonnes of CO_2 so is significant compared to potential GHG benefits. There is also concern about potential increase of soil erosion problems on land when cereal residue always removed. Finally, there are concerns about the nutrient removal with residue and effect on future fertility needs. More work is warranted to include for total life-cycle environmental effect of straw harvesting.

Effect on deforestation?

The role of deforestation is important on a worldwide scale and several analysis have indicated biofuel production will increase such clearing with huge GHG releases such that no net GHG benefit from biofuel produced on agricultural land (Farigone *et al*. 2008; Searchinger *et al*. 2008). Although the major concern is deforestation of tropical forest, over the last 20 years there has been about 30 to 80 thousand hectares of annual clearing of forests to increase agricultural land in Canada. This occurs both as clearing of wooded areas within agriculturally developed areas and as clearing of natural forest at the frontier between agriculturally developed and unsettled areas. Under an aggressive bioenergy scenario, agricultural land prices are predicted to increase 40 to 60% in western Canada and up to 200% in eastern Canada (latter where maize is best suited). Therefore, the land price increase will be an encouragement to clear trees as alternative to buying existing agricultural land for farm expansion. Since GHG emissions for clearing average about 200 tonne of CO_2 equivalent per ha (mostly from the loss C in the trees themselves), about 95 thousand hectares of additional deforestation per year would eliminate the GHG from an aggressive bioenergy production policy. Such increased deforestation rates are feasible as we estimate there are 6 million ha of land currently under trees in Canada with good capability for arable agriculture. Those emissions would be reduced greatly if the woody biomass were used as a lignocellulosic feedstock for bioenergy to replace fossil fuels. However, an expanded energy demand for woody biomass would also provide additional incentive to deforest as it provides a market for the cleared woody biomass that, in many cases, currently has no market value and is simply burned in field piles as a disposal method.

In addition to concern about increased deforestation in Canada, there is also concern that less net foodstuff exports from Canada due to bioenergy production could contribute pressure for deforestation to agriculture in other countries as they try to increase their food production capability to meet human demand.

Conclusion

Aggressive bioenergy production using feedstocks from existing agricultural land in Canada does increase rural development. Society will have to address the potential competition between food production and biogenic energy as economics could favour energy production over food. There are also important GHG benefits from the substitution of biogenic energy produced on agricultural land for fossil fuel energy. More complete analysis that includes the effect of potential deforestation resulting from bioenergy production on agricultural land and from residue removal is needed to fully assess the environmental benefits of bioenergy production from agricultural land.

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