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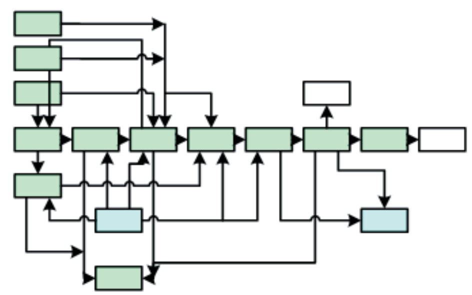
DIAS report • Animal Husbandry • no. 61 • October 2004 • Life Cycle Assessment in the Agri-food sector

DIAS report

Life Cycle Assessment in the Agri-food sector

Proceedings from the 4th International Conference,
October 6-8, 2003, Bygholm, Denmark

Niels Halberg (ed.)



Ministry of Food, Agriculture and Fisheries
Danish Institute of Agricultural Sciences

DIAS report • Animal Husbandry • no. 61 • October 2004

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Danish Institute of Agricultural Sciences Department of Agroecology

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Preface

Focus on sustainable (agricultural) production should go hand in hand with the quest for sustainable (food) consumption. This idea is in agreement with principles of Integrated Product Policy, which is the modern chain based and product-oriented approach to environmental improvement and regulation adopted by national environmental authorities, the EU Commission, UNEP and increasingly also by private companies. The regulation of agri-environmental problems has so far focused on improvements in farm production systems in relation to local effects. This approach does not take into consideration the developments in food processing and consumer choices as well as more global environmental effects. In the coming decades production and consumption of livestock products and processed food is expected to increase significantly and the globalisation of the food systems will continue. Therefore, as much as ever there is a need for a complementary focus on the environmental impact per unit of food produced including the whole production chain and taking into account global effects of different systems for food provision and consumption patterns.

Food consumption is one of the major causes for resource use and environmental impact by modern households. But the relative importance of these burdens in the primary production, industrial food processing and kitchen preparation respectively differ among products. It is, however, not easy to define and compare the environmental burden from different choices of food. Life Cycle Assessment (LCA) is a tool for an aggregated description of emissions, waste and the resource use from soil to kitchen per unit of different food items. The methodology for LCA in the food sector has been developed during the last decade and progress in terms of methodological robustness and data availability has been demonstrated - among others – at a series of LCA-food conferences organised in Brussels (1996, 1998) and Gothenburg (2001). The present proceedings from the Fourth International Conference of LCA in the Agri-Food Sector presents a number of papers based on floor- and poster presentations showing both the recent advances in methodology and inventories and the wide range of applications and objectives for LCA in the food sector.

The first 10 papers and 10 of the 21 poster papers demonstrate different applications of LCA in the primary sectors agriculture, horticulture, livestock production and aquaculture/fisheries and in the food processing industry. Then follows a section of papers and posters addressing methodological questions - such as system expansion and land use - and presenting new inventories of life cycle data relevant to the food sector. Finally, a number of papers and posters present other approaches to sustainability assessment of food production and consumption, which may supplement the more “classical” LCA.

The organisers wish to thank the authors for providing revised papers for these Proceedings and the referees for giving constructive advise for improvements of the full papers and poster papers.

Niels Halberg

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Moreover, the organizers are grateful for help from the programme committee and the following companies/organizations: 2.-0 LCA consultants, UNEP, Vito and SIK.

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The papers in this volume and the presentations at the conference are available at the website: http://www.lcafood.dk/lca_conf/

Life cycle assessment of bread production - a comparison of eight different scenarios -

Braschkat, J.[1]), Patyk, A.[1]), Quirin, M.1), & Reinhardt, G. A.[1)]*

*[1)] ifeu - Institut für Energie- und Umweltforschung Heidelberg GmbH, Wilckensstr. 3, D-69120 Heidelberg, * Tel: +49-6221-4767-0 (ext.: -31), Fax: -19,* guido.reinhardt@ifeu.de

Abstract

In a study the life cycle assessments of eight different ways of bread production were evaluated, considering different crop production methods (conventional, organic), different milling technologies (industrial mill, domestic mill) and different baking technologies (large bread factory, bakery, domestic bread maker). The scenario combining organically grown wheat, industrial milling and a large bread factory proved to be most advantageous way of producing bread with respect to the impact categories considered in this study. The use of organically grown wheat, however, requires more land area than the use of conventionally grown wheat. In addition to the differences due to the crop production method, milling and baking technology, the transport of grains, flour and bread by the consumer is of vital importance for the ultimate appraisal of each scenario. In fact, the transport by the end consumer easily dominates the overall ecological effect depending on the distance.

Keywords: life cycle assessment, bread, agriculture.

Introduction

Bread is one of the most important foods. Like any other production process, bread production is associated with environmental impacts due to the demand of resources and due to emissions. The environmental impacts, however, vary depending on the way bread is produced. Disseminating information about the environmental effects of different ways of bread production would enable the consumer to base his decision of purchasing bread or its ingredients on ecological reasons. Bread producers on the other hand might use this information to optimise the production process. In order to provide a complete image of all relevant environmental aspects such as resource and energy demand, greenhouse effect, ozone depletion, acidification, eutrophication, photo smog and demand of land area, the entire life cycle from the acquisition of raw materials, across the actual production and use, up to the final disposal/recycling, needs to be considered. Life cycles within bread production may differ already during the production of the wheat, which may be grown either conventionally or organically. After crop production, milling of the grain might be done either by an industrial mill or alternatively by a domestic mill. Finally, bread might be produced from the flour either in a large bread factory or in a bakery or in a private household (bread maker). Each of these single options is again associated with different transport efforts, which have to be considered as well.

This study based on Patyk (2003a) and on own calculations aimed at answering the following questions: Which way is the most environmentally friendly way of bread production? And, more specifically: Which crop production method (conventional, organic), which milling technology (industrial or domestic mill) and which baking technology (large bread factory, bakery, domestic bread maker) is the most advantageous one from the ecological point of view? Which one of the process steps, including the transports, does account for the highest or lowest environmental effects? Where, within the whole bread production chain, is it feasible to introduce ecological optimisations or to reduce environmental effects, and what are the corresponding recommendations?

Approach and predefinitions

The combination of the above mentioned single processes result in 8 different scenarios of bread production (Figure 1) like, for instance, bread made from organically grown wheat, ground and baked in a private household including the associated transports. The life cycles assessed in this comparison start off with the crop production, incorporating all steps of the conventional as well as organic wheat production, from soil cultivation up to harvest. Data for soil cultivation, like ploughing, seedbed preparation, sowing, application of fertilisers and pesticides (conventional production only), harvest, transport of the grain to the farm and seed production was taken from Borken et al. (1999). Data for the provision of diesel fuel are based on Patyk (2003b). For fertiliser production data from Patyk & Reinhardt (1997) and for additional operational supplements data from Gaillard et al. (1997) and Reinhardt et al. (2001) were used. For the conventional as well as for the organic system the entire crop production was considered within the complete crop rotation. Consequently, various effects like the value of preliminary crops or intercropping for nitrogen fixation in the organic system, resulting in different yields per unit area and time, were implicitly taken into account (for details we refer to Kaltschmitt & Reinhardt (1997)).

Within flour production it is assumed that the grain is ground either in an industrial mill or in a domestic mill. Industrial mills are operated at medium voltage; domestic mills in contrast are run at low voltage. In this study, data for the provision of electrical energy given in Borken et al. (1999) and data for the energy demand of industrial mills from Fritsche et al. (2002) were used. The energy required for operating domestic mills and for the provision of operating supplies within this study, was based on estimates.

Regarding bread production three different options were considered: a large bread factory, a bakery and a private household using a bread maker. Ovens used in bread factories and in bakeries are operated either by electricity or oil or natural gas while domestic bread makers are solely driven by means of electrical power. For the commercial ovens the energy-mix according to Fritsche et al. (2002) was applied. The energy demand of domestic bread makers was estimated and linked with the data on energy supply published by Borken et al. (1999) and Patyk (2003b).

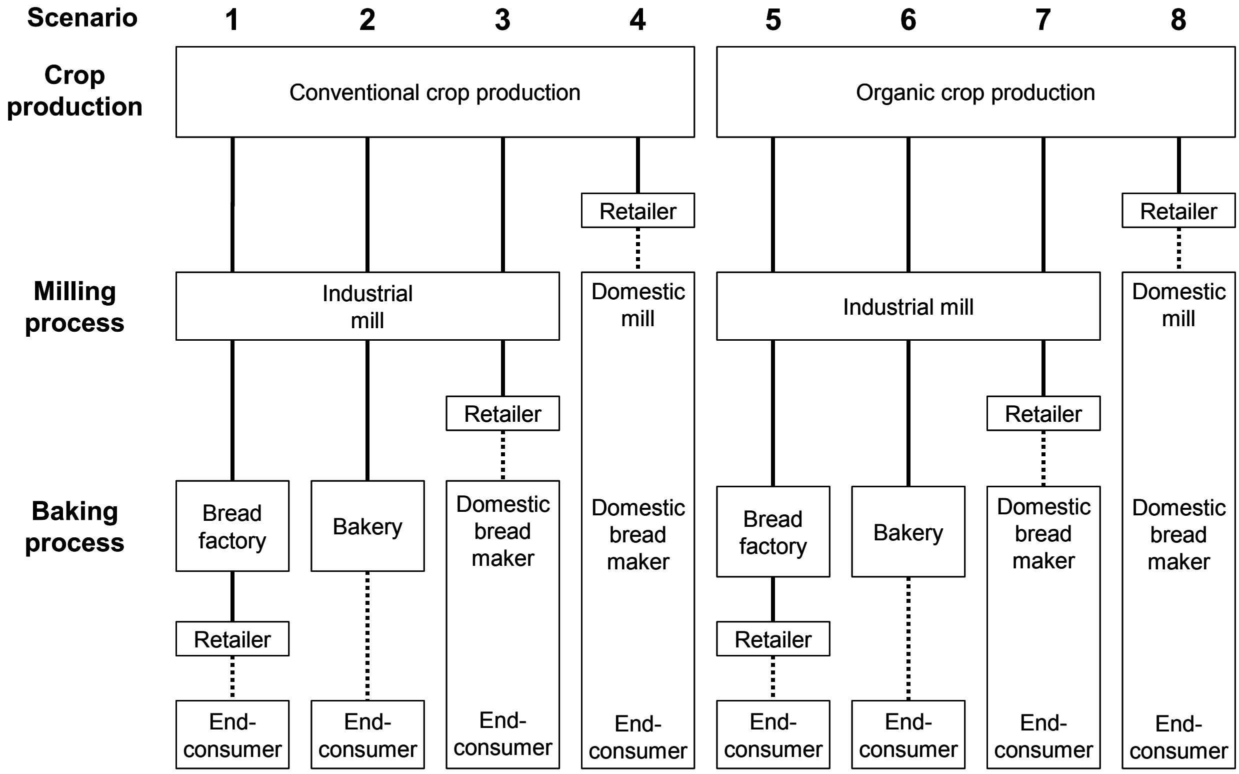

Figure 1: Schematic representation of the 8 life cycle scenarios of bread production (solid lines indicate standard transports, broken lines indicate transports by consumer).

Standard transports with 23 t trucks (distance 100 km, outward bound fully loaded, return empty) were assumed for the following routes: transports of grain from the farm to the mill or retailer, transports of flour from the mill to the bread factory or bakery or retailer and transports of bread from the bread factory to the retailer. Data on diesel fuel consumption for these transports originate from Borken et al. (1999) and Knörr (2002). Data on the provision of diesel fuel and operating supplies were taken from Patyk (2003b) and Reinhardt et al. (2001) respectively. For the transport of grain, flour and bread by the end-consumer it was initially assumed, that the transports were done either on foot or using a bicycle and thus, the energy demand and emissions were either zero or not significant. Finally, additional scenarios for the transport of the bread by the consumer using a car were calculated.

Basically, the assessment follows the ISO 14040/43 standards. The functional unit is 1 kg of bread ready for consumption at home. The environmental effects studied are listed in Table 1 including cumulated primary energy of non-renewable energy carriers, greenhouse gas emissions by IPCC method, ozone depletion through N_2O emissions, eutrophication and acidification through airborne emissions and others. For details we refer to Borken et al. (1999).

Table 1: The environmental effects, indicators and parameters considered in this study.

Environmental effect	Indicator	Parameter
Energy demand	Non-renewable primary energy	Crude oil, natural gas, mineral coal, lignite, Uranium
Greenhouse effect	CO_2-equivalents	CO_2, N_2O, CH_4
Ozone depletion	N_2O	N_2O
Acidification	SO_2-equivalents	SO_2, NO_x, NH_3, HCl
Eutrophication	PO_4-equivalents	NO_x, NH_3
Photo smog	Ethen-equivalents	CH_4, NMHC
Land use	Land use	Land use

Results

The environmental effects (Table 1) of the 8 different scenarios of bread production are presented comparatively in Figure 2. Values refer to the production of 1 kg of bread. The baking process was the most energy-consuming step of the entire bread production process accounting on average for 64% of the total energy demand. The baking process using a domestic bread maker requires 3 times more energy than in a factory and in the bakery, energy demand is still twice as high than in a large bread factory. Due to the close correlation between energy demand and greenhouse effect the same applies to the greenhouse effect as well. Besides, using a conventional oven for baking bread at home requires more energy on average than a bread maker and therefore this option was not considered in this study.

Crop production, however, is much more important regarding the greenhouse effect because of the amount of N_2O released and for that reason the assessment not only depends on the baking process but also on the way, how the crop was produced. Regarding ozone depletion, acidification and eutrophication the situation is completely different. In these cases, all of the scenarios based on organic crop production are most beneficial, whereas the remaining downstream processes did not entail any further differentiation of the results (Figure 2). Regarding photo-smog, ethene-equivalents (NMHC) as well as the NO_X-corrected ethene-equivalents (NCPOCP) have been considered. However, this analysis did not show any significant differences between the eight scenarios.

From the environmental effects considered so far, only advantages resulted from organic production of the wheat crop. This situation changes when considering the size of the land area that is required for the crop production. The conventional production system requires only 65% of the area that is needed to grow the wheat organically (Figure 3). This is largely due to the use of synthetic fertilisers and pesticides and the resulting higher yields in conventional farming systems.

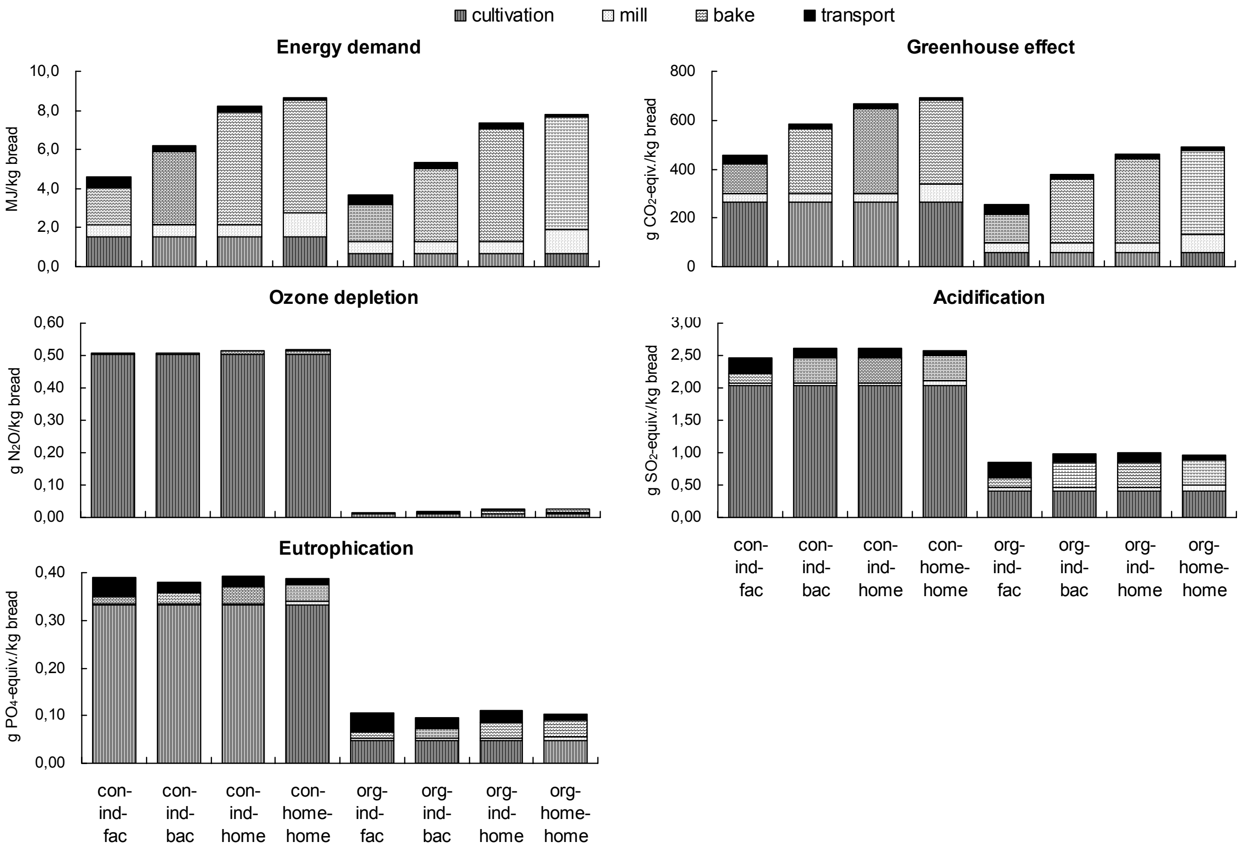

Figure 2: LCA results of the 8 scenarios of bread production regarding energy demand, greenhouse effect, ozone depletion, acidification and eutrophication.

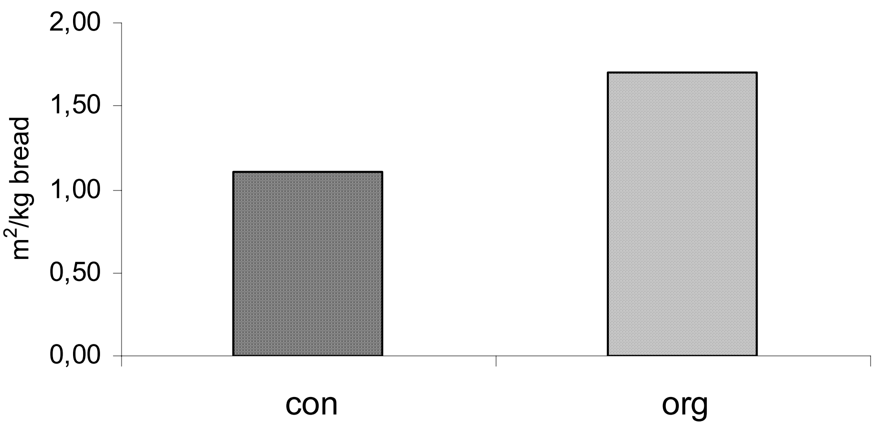

Figure 3: Land area required to produce the grain for 1 kg of bread using a conventional (con) and an organic (org) production system.

For the transport of grain, flour and bread by the consumer different transport modes (car, public transport, bicycle/on foot), distances and additional transport purposes have to be considered. In an additional analysis, the ecological impacts were calculated for a number of different transport scenarios, in order to demonstrate the magnitude of the effects, owing to the transport of 1 kg of bread by the consumer. Exemplary results are presented in Figure 4 for energy demand and acidification. In contrast to the transport on foot or by bicycle, which involves zero, respectively negligible emissions and demand of energy, the extreme case of a 4 km car transport, with the sole purpose to acquire 1 kg of bread, will entail an energy demand of 18.6 MJ and the emission of 2.2 g SO_2–equivalents. That means the energy demand due to the bread transport is 4 times higher and acidification is 2.5 times higher than that caused by the entire preceding bread production chain (conventional crop production assumed). In addition to this extreme transport scenario, Figure 4 indicates car transport distances, at which the higher energy demand caused by a domestic bread maker and by a bakery, in contrast to a large bread factory, was compensated. Similarly, the car transport distance is indicated, at which the larger acidification due to the conventional crop production was compensated. As a result a transport distance of about 1 km and 0.5 km may compensate for the higher energy demand of a domestic bread maker and of a bakery compared to the bread factory. Larger transport distances will overcompensate the higher energy demand of the bread maker and the bakery. In contrast, the acidification due to the conventional crop production compared to the organic crop production will be compensated at a distance not less than about 3 km and that again only if the bread purchase is the sole purpose of the drive. Adding another 10 kg of groceries to the shopping will increase the distance, at which the higher acidification of the conventional crop production is compensated, up to 29 km.

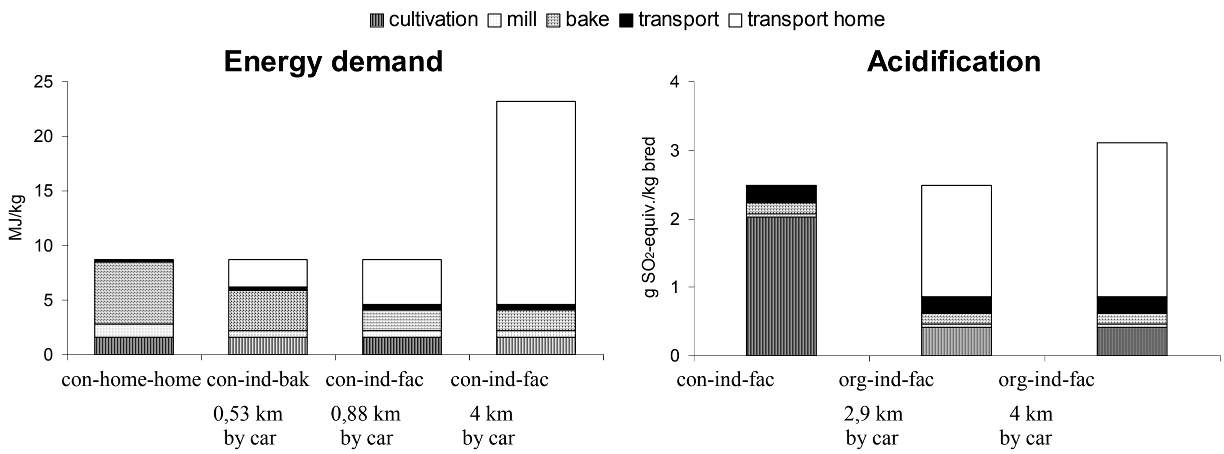

Figure 4: Magnitude of the energy demand and the acidification due to the transport by the consumer in addition to the production of 1 kg of bread.

While the larger energy demand and the higher greenhouse gas emissions of a bread maker in contrast to a bread factory is compensated by a 1 km car transport, the car transport will cause a less favourable result for the bread factory in the remaining categories. However, this implies that the grain or flour for a bread maker was transported either on foot or by bicycle.

Conclusions

Looking at the single options of each processing step (crop production, milling, baking) the following conclusions may be drawn: Organically grown wheat has to be preferred over wheat that was produced conventionally, regarding all impact categories except land use. Flour may be produced preferably in an industrial mill rather than in a domestic mill. Ranking the bread baking process from the most to least advantageous option results in the order bread factory, bakery and domestic bread maker.

Looking at the entire bread production chain including the transports by the consumer the following conclusions may be drawn: Bread production using organically grown wheat, ground in a industrial mill and baked by a large bread factory is the most preferable way of producing bread. As far as possible bread producers may have to use cereals originating from organic production. Transport of grains, flour and bread by the consumer is of vital importance for the ultimate appraisal of each scenario. For instance, concerning the energy demand, the consumer annihilates the entire ecological advantage of the bread factory, if he involves a transport by car over a distance of just 1 km. As a principle consumers may have to ask for eco-bread. Bakery products should generally be transported either on foot or with a bicycle. If transported by car, the purchase of bread may be combined with the shopping of additional groceries. A bakery might look for commercial energy saving measures. For instance, participating in projects like "Bäcker/Konditoren und Umwelt" organised by the bakery guild, the BUND and the city council Heidelberg may be a first step (Bäckerinnung et al. (2003)). Homemade bread may be baked in a domestic bread maker because the energy demand for the production of 1 kg bread is lower than the production in a domestic oven. If using the oven anyhow, increasing the degree of utilisation may reduce the energy demand of the domestic oven.

For the decision of a specific source of bread supply not only ecological aspects may be relevant but also socio-political aspects (promotion of small enterprises), economic aspects, nutritional aspects, recreational aspects as well as hobby aspects play an important role. Therefore, the selection of a specific bread supply option is based on the individual choice of all these issues.

References

Borken, J., Patyk, A. and Reinhardt, G. A. 1999. Basisdaten für ökologische Bilanzierungen: Einsatz von Nutzfahrzeugen in Gütertransport, Landwirtschaft und Bergbau. (Basic data for life cycle assessments: Use of commercial vehicles in transportation, agriculture and mining). Verlag Vieweg, Braunschweig/Wiesbaden

Bäckerinnung, Bund für Umwelt und Naturschutz (BUND) and Stadtverwaltung Heidelberg 2003. Projekt "Bäcker/Konditoren und Umwelt" (Bakery and Environment). Heidelberg, running project

Fritsche, U. R. et al. 2002. Gesamt-Emissions-Modell Integrierter Systeme (GEMIS) (Global Emission Model for Integrated Systems (GEMIS)) Version 4.1

Gaillard, G., Crettaz, P. and Hausheer, J. 1997. Umweltinventar der landwirtschaftlichen Inputs. (Environmental inventory of agricultural inputs). FAT-Schriftenreihe Nr. 46, Tänikon/CH

Kaltschmitt, M. and Reinhardt, G. A. (ed.) 1997: Nachwachsende Energieträger: Grundlagen, Verfahren, ökologische Bilanzierung (Biofuels: Basics, approaches, environmental assessments). Verlag Vieweg, Braunschweig/Wiesbaden

Knörr, W., Höpfner, U., Lambrecht, U., Nagel, H.-J., Patyk, A. 2002. Data and Calculating Model: Energy demand and pollutant emissions from the transportation sector in Germany 1960 to 2020. Development of the software TREMOD by IFEU-Institute Heidelberg, Data-preparation and calculations by order of the Federal Environmental Office, Berlin. Documentation

Patyk, A. 2003a. Inventory Analysis of Petroleum Products. By order of the German Scientific Association for Crude Oil, Natural Gas and Coal, supported by the Association of the German Petroleum Industry, Hamburg, in print

Patyk, A., 2003b. Der KEV im Bereich Landwirtschaft/ Nahrungsmittelproduktion - Vergleich zur Herstellung verschiedener Brotarten. (CED in agriculture and food production - comparison of the production of different bread types). Published in the final report to the project "Anwendung und Kommunikaton des Kumulierten Energieverbrauchs (KEV) als praktikabler Entscheidungsparameter für nachhaltige Produkte und Dienstleistungen". (Utilisation and communication of the Cumulated Energy Demand (CEV) as a practical decision parameter for sustainable products and services)

Patyk, A. and Reinhardt, G. A. 1997. Düngemittel – Energie- und Stoffstrombilanzen. (Fertilizer – energy and production flow anlasysis). Verlag Vieweg, Braunschweig/Wiesbaden

Reinhardt, G. A., Jungk, N., Korsuize, G. and Patyk, A. et al. 2001. Bioenergy for Europe: Which ones fit best? – A comparative Analysis for the Community. By IFEU/D (coordinator), BLT/A, CLM/NL, CRES/GR, CTI/I, FAT/CH, INRA/F, TUD/DK. FAIR CT 98 3832.

When a hole matters - the story of the hole in a bread for French hotdog

Rosing, L. and Nielsen, A.M.

**) Cerealia Bakeries A/S, Birkemosevej 10, DK-8361 Hasselager. Tel.: +45 76 28 50 00. Fax: +45 76 28 50 01. lrg@unibake.dk. 2.-0 LCA consultants. Borgergade 6, 1., DK-1300 Copenhagen.*

Abstract

The environmental management of Cerealia Bakeries has till now focused on the bakeries' own consumption of energy, packaging and cleaning agents. However, looking at the entire chain of processes, which are affected when a consumer eats industrial bread products, the major environmental impacts arise from actions by the suppliers and customers. In this paper, the environmental impacts of these production technologies are assessed, and the implication for future product development is discussed.

Background

During the last years, Cerealia has participated in development and application of product-oriented environmental assessment of food. This paper is based upon an environmental assessment of improvement potentials in the production of bread for French hotdog (Rosing and Nielsen, 2003), which was made as part of the LCAfood-project. Cerealia has published part of their own environmental data in the LCAfood-database (Nielsen et al., 2003), which in return has been the source of data on environmental data from agricultural productions for Cerealias work.

Identification of environmental hot-spots in the product chain of bread for French hotdog

Previous experiences show us that the hot-spots of a roll lie in agriculture's production of grain, including their use of fertiliser, and the consumers' use of oven for heating up the bread (Rosing et al., 2001). As part of the LCAfood-project, the environmental impacts from bread for French hotdog was analysed, and a similar picture was found.

Figure 1 shows the environmentally most important processes, which are affected when a consumer buys a bread as part of a French hotdog, and their contribution to global warming potential, calculated with the EDIP-method (Wenzel et al., 1997). The chosen amount (17.000 pieces of bread sold from hotdog-stand) equals the product in one hour in Cerealias production unit in Karup (18.000 pieces of bread sold from bakery minus waste in the hotdog stand). The breads are produced of conventional wheat flour, distributed to the hotdog-stands, where they are heated by use of a toaster. The by-products from the milling-process are used as animal feed, and thus replace alternative feed (barley). This decreased production is shown as the green bar.

More than 50% of the total contribution to global warming comes from agriculture's production of grain. For the other impact types eutrophication, acidification and nature occupation (land-use), this percentage is even higher.

Use of conveour toaster contributes with 25% to the total global warming. This percentage can vary between 10 – 60% according to the size of the hotdog stand and how fast the hotdogs are sold.

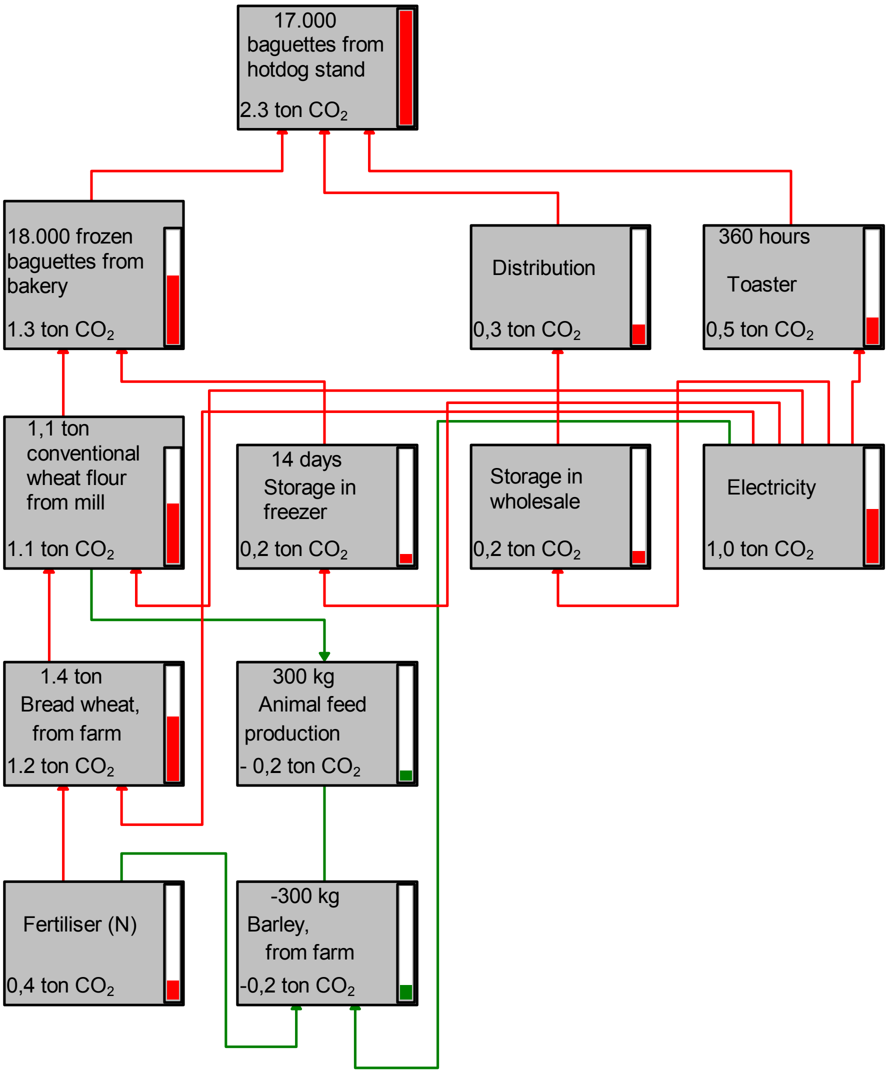

Figure 1. The product chain for bread for French hotdog sold at hotdog stand. Only processes, which contribute with more than 6% to total result are shown. Boxes refer to production processes. Names of grey boxes refer to the main product of the processes. Red arrows represent material or energy transfer between two processes; green arrows represent saved material or energy transfer as a result of displacements and green lines represent displacements. The red/green-bars in the boxes show each process' relative, cumulated contribution to total environmental impact.

Different production-technologies for making the hole in the bread for French hotdog and their environmental impact

A baguette for French hot-dog must have a hole to put the sausage into. Until 15 years ago, all hotdog stands bought standard baguettes, and made the holes themselves by sticking a thick spear into them. This method was not without pitfalls. Partly because a big amount of bread was pressed together in the bottom of the hotdog, partly because some hotdog stands had to throw big amounts of baguettes out, when they destroyed them by e.g. sticking the spear out of the side of the baguette.

Some hotdog-men suggested to Cerealia to find a better way of making this hole. Then the present production technology was invented, where the baguettes after baking are chilled, frozen, and drilled. In the beginning, the bread waste was sent to combustion with other normal waste. To avoid the big amounts of waste, it was tried to sell the fresh bread-waste as animal feed. The distribution was difficult, because it was necessary that the bread should be sold and used within two days to avoid mould.

Cerealia then started using the bread-waste for making dried breadcrumbs. Today, all bread waste from the drilling of breads for French hotdogs is used in this production. The breadcrumbs are sold to household-consumers, to other food industry, or reused in-house for production of new baguettes, where they replace flour. Excess amounts of breadcrumbs are sold to farmers as animal feed, where they replace alternative feed (barley).

From this story we can identify three production-technologies for bread for French hotdogs, and as a future outlook add a forth:

A. The breads are frozen and drilled. The bread waste is sold (fresh) as animal feed, where it replaces barley.
B. The breads are frozen and drilled. The bread waste is processed into breadcrumbs, and used in the bakery in stead of flour.
C. The breads are frozen and drilled. The bread waste is processed into breadcrumbs, and sold as animal feed, where it replaces barley.
D. A future best-case technology, where bread-waste from drilling can be avoided. Either the breads are baked around a stick, or a new recipe is developed where the breads can be baked with more air in them, so the hotdog-man can make the hole with a spear, still avoiding the excess amount of bread in the bottom of the hotdog.

In Figure 2, the contribution to environmental impacts is shown for a shift from the production-technology 15 years ago, where the bread was sold at a weight of 90 g, and the hole was made by the hotdog-man with a spear, to any of the four above-mentioned technologies.

Figure 2 should be read like this: if a hotdog-man changes from ordinary baguettes to baguettes with a pre-drilled hole in them (C), the contributions to all studied environmental impacts will decrease. The environmental impacts will decrease even more, if the bread is used as feed without the drying of breadcrumbs, or if the dried breadcrumbs can be used as an alternative to flour in another production. A hypothetical, new technology where the breads can be produced without any drilling and/or drying of breadcrumbs, would lead to the biggest environmental improvement (D).

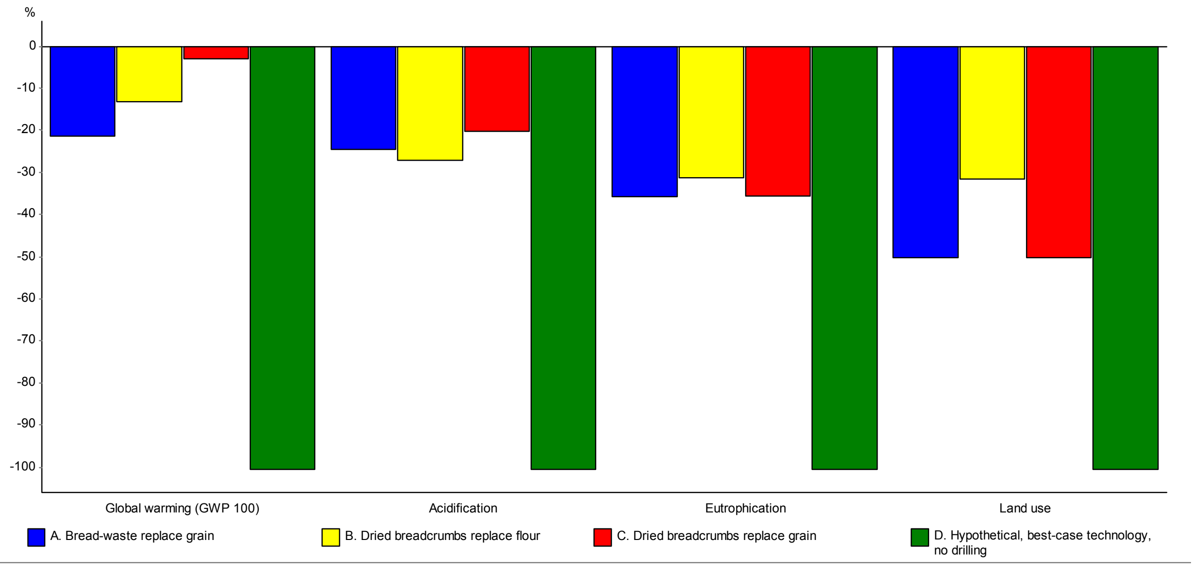

Figure 2. Comparison of environmental impacts (Global warming (GWP 100), Acidification, Eutrophication, Land use) from a change in production-technology from old production, where the hole is made (A. Bread-waste replace grain. B. Dried breadcrumbs replace flour. C. Dried breadcrumbs replace grain. D. Hypothetical, best-case technology, no drilling). Negative bars shows that a change to the technology yields an environmental improvement.

Figure 2 shows that the last years' shift in production-technology has not just made the hotdog-men more satisfied with the breads for French hotdogs, but has also been an improvement for the environment. All the bars show decreased environmental impact, meaning that a shift to any of them will be an environmental improvement. Even though the energy-consumption at the bakery has increased, because drying of breadcrumbs is an energy-consuming process, the total environmental impact is decreased, because even more energy-consuming processes are avoided in the production of grain or flour.

In A and C the bread waste from the drilling replaces the same amount of alternative feed in agriculture. When the bread-waste from the production is used as animal feed, the same amount of grain is replaced as animal feed, and thus the same amount of land will not have to be cultivated, whatever the bread is processed into dried breadcrumbs, or sold as fresh bread. But in alternative A, where the drying of the bread is avoided, will off course have a lower consumption of energy, and thereby a lower contribution to global warming, acidification and eutrophication.

The present production-technology (C) where the hole is drilled at the bakery, and the bread-waste is processed into dried breadcrumbs has the largest environmental impacts of the studied alternatives. Therefore a change to any of the alternative technologies could lead to environmental improvements.

Discussion

Cerealias work with LCA has given a new basis for prioritizing future working efforts in the environmental department. The hot-spot identification has pointed to agriculture and hotdog-stands as key processes, which would be important to change to lower the total environmental impact, but the production of bread for French hotdogs shows that environmental improvements can also be found through development of the bakeries' own productions.

Cerealias opportunity to influence agricultural production

The majority of environmental impacts from the bread come from agriculture's production of wheat. Based on the Danish data from the lca food database the present supply of flour-types offers Cerealia no possibility to replace their use of conventional flour with a less environmentally problematic flour-type. The alternative flour-types could maybe have lower toxic impact, e.g. organic flour is produced with no use of pesticides or more environmental friendly flour e.g. without straw shortener. However, the contribution to global warming, acidification and nutrient enrichment will be similar for them, and if the flour comes from agriculture with a lower productivity, the impact on nature occupation can even increase. One of Cerealias environmental goals is a.o. to increase the use of more environmental friendly flour, why we in the future will do some analysis of different flour types to examine the alternatives.

On the other hand, Cerealia Bakeries have a possibility to decrease environmental impacts from agricultural production by decreasing the demand for grain through utilising of bread-waste. The development in production-technology of bread for French hotdog is an example of this in two respects:

1. bread-waste is used as a replacement of flour or grain
2. new types of bread are developed, where the customers needs are fulfilled with the lowest amount of bread

When possible, Cerealia participates in research aimed at lowering the environmental impacts from agricultural production. Knowledge from these projects will continuously be included in the company's decision-basis.

Cerealias opportunity to influence hotdog-men

The hotdog-stand can be the second most important process in the environmental impact of the bread. If the hotdog is sold from small or inefficient hotdog-stands, it may even exceed agriculture in contribution to the environmental impacts.

Cerealia has a clear possibility to influence the preparation routine of the hotdog-men, since guidelines for preparation of the breads are part of normal marketing procedure. If Cerealia find concrete suggestions to improve the environmental impact from the hotdog-stands, such information could be included. However, consumers are not likely to care much about the environmental impact of the hotdog-breads, and hotdog-men are not likely to work for decreased environmental impacts from their products, unless it is economically and practically feasible. But seeing that energy-savings and lower amounts of bread-waste mean lower cost for the hotdog-man, this should be possible.

In contrast to the agriculture, there are no research-projects on environmental impacts from hotdog-stands that Cerealia can join.

Cerealias opportunity to achieve environmental improvements by changes in the bakery
Organizationally it is easier for Cerealia to change the production in the bakeries, than to influence the behaviour of suppliers or customers. As showed in this analysis the absolute value of the environmental improvements is a change in production technology of the hole. This is not an option for now, as our customers are happy with the product. Therefore we have instead looked at other possibilities within the bakery. During our environmental project "LCA – as a tool in the learning bakery" we have created working groups which should analyse the environmental improvements at the bakeries. Another lca could maybe show the positive decrease of environmental inpact for the bakery, but this work has not been done.

References

Nielsen, P.H., Nielsen A.M., Weidema B.P., Dalgaard R. and Halberg N. (2003). *LCA food database*. www.lcafood.dk/.

Rosing L, Feldvoss C, Kann M, Leth K D, Mundt P, Pedersen H, Skovsby K, Christensen L M. (2001). *LCA of a roll* (in Danish). Working report in the project LCAfood. Available at www.lcafood.dk.

Rosing L and Nielsen, A.M. (2003). *Bread for French hotdog – possible environmental improvements in the product chain.* Available at www.lcafood.dk.

Wenzel H, Hauschild M and Alting L (1997). *Environmental assessment of products*. Volume 1. Methodology tools and case studies in product development. Chapman and Hall.

Possible benefits from using LCA in the agro-food chain, example from Arla Foods

Inger Larsson

Arla Foods, Group Environmental affairs, Sweden, tel: +46 8 6773277, mobil: + 46 705 145640, fax: +46 8 789 53 20

Arla Foods has worked with life cycle assessment since the mid 90s. We have worked with LCA concerning packaging, transports and different milk products. The work with LCA for packaging and transports was originally based on the public interest from consumers concerning the environmental impact from packaging and transportation. The work has then developed to become complete LCA for different milk products. A life cycle assessment has, in cooperation with the Swedish dairy association, been made for drinking milk packed in Tetra Brik packaging. The work has been published as a scientific report and has also been published as a brochure that can be found on www.svenskmjolk.se/english.asp Download: Milk and the environment. The life cycle assessments have given us basically five kinds of information to date. The production of milk at the farm including production of concentrated feed, fertilizers, pesticides etc., is without doubt the most important impact on the life cycle of milk and cheese. The most important environmental work for liquid milk dairies is to reduce the loss of milk to wastewater, pig feed and landfill. For cheese dairies primary work is to produce a high cheese yield.

- *Transports of milk in tankers are efficient and have only little impact on the environment when counted per liter of milk.*
- *The transport of milk or cheese to the home can cause substantial environmental effect.*
- *The choice of packaging material has an important influence on the environment. A carton material is always better than plastic ones. Whether package material is recycled or incinerated makes less difference than the choice of material.*

Using LCA results, Arla Foods has now advanced environmental work in the following ways.

- *We can give better answers about environmental issues to consumers, customers and students.*
- *New LCA information is published in brochures and reports.*
- *The liquid milk dairies in Sweden have set targets to reduce all kinds of milk loss. These dairies also have targets to avoid increasing the environmental impact of packaging.*
- *The environmental work includes the Arla Foods farms in Denmark and Sweden. The quality program, including the environmental program for Arla Foods farmers in Sweden and Denmark is available in Swedish and Danish, log on to www.arlafoods.dk for more information, click download "Arlagården".*

Environmental Management Practices in an Italian Coffee Company using LCA Methodology

Roberta Salomone
Dipartimento RIAM - Università di Messina - Piazza S. Pugliatti 1, 98121 Messina
Tel. 0039-90-771548; 0039-90-661499, fax 090/6764920, e-mail: roberta.salomone@unime.it

Abstract

Coffee production has grown by nearly 200 percent since 1950; it is the most important traded commodity in the world (after oil). Even if it is cultivated only in tropical and equatorial areas (is the primary export of many developing countries), the most coffee is consumed in the developed world (the United States and the European Community together import two out of every three bags of coffee produced in the world). Considering that the coffee chain is very wide and interspatial, with the involvement of many companies of different type and size, each environmental decision, in whatever point of the coffee chain, should be taken in a "life cycle thinking" perspective. It was with this intention that an environmental analysis of impacts connected to a coffee industry located in Sicily was conducted, applying Life Cycle Assessment methodology. System boundaries were defined to include all life cycle steps from coffee cultivation to the consumer distribution, consumption and disposal and the functional unit was defined as 1 kg of packaged coffee delivered to the final consumer. The impact assessment step was performed investigating eight different impact categories (Air acidification, Aquatic Eco-toxicity, Eutrophication, Human toxicity, Terrestrial Eco-toxicity, Greenhouse effect, Depletion of ozone layer, Photochemical oxidant formation) and Eco-points were used as a general weighting factor. The cultivation and the consumption stages were identified as having the greatest environmental impacts. The aim of the study was to recognise the "hot spots" in the product's life cycle stages, in which environmental improvements are easily achievable and to suggest options to minimise the environmental impact of their production phases, improving process and company performance. The results showed that the LCA methodology is suitable to assess the environmental impact associated with the entire life cycle of an agro-food product and also to improve gate-to-gate life cycle information that provide valuable means of understanding at company level; this means that environmental improvements could be taken in a "life cycle thinking" perspective.

Introduction: aim of the study

Coffee production has grown by nearly 200 percent since 1950; after oil it is the most important traded commodity in the world. Although it is grown only in tropical and equatorial areas (it is the primary export of many developing countries) most of the coffee produced is consumed in developed countries (the United States and the European Community jointly import two out of every three bags of coffee produced in the world) (www.wri.org). Considering that the coffee chain is very wide-ranging, involving many different types of company of varying size, each environmental decision, at any point whatsoever of the coffee chain, should be taken using "life cycle thinking".

Environmental management has become increasingly important to productive and innovative businesses and often involves suppliers upstream and the companies downstream. A business that wishes to implement an effective internal environmental management system must first of all analyse the environmental impacts of its production process and its products/services. Inevitably, this entails identifying impact factors found at the start or end of pipeline and therefore outside the physical confines of the business' own productive sphere of activity (Mirulla, 2001). Life Cycle Assessment (LCA) is making its mark as one of the most interesting tools available to management for environmental assessment and control. LCA broadens the vision of a producer giving it a more generalised view of the environmental impacts of the production line. The business has to involve suppliers upstream and the companies downstream to collect inventory data in accordance with the boundaries of the system analysed.

This paper presents an environmental analysis of a coffee business adopting LCA methodology. The analysis was carried out on a firm in Sicily (Italy) that roasts and distributes coffee. The aim of the study is to obtain data relating to energy use, waste management and raw material consumption in order to identify the "hot spots" in the stages of the product's life cycle in which environmental improvements are easily achievable then to suggest alternatives to minimise the environmental impact of production phases, thereby improving processes and company performance.

Methodological framework

The analysis of the environmental impacts of a coffee company located in Sicily was investigated by applying Life Cycle Assessment (LCA). LCA is a methodology used for analysing and assessing the environmental loads and potential environmental impacts of a material, product or service throughout its entire life cycle, from raw materials extraction and processing, through manufacturing, transport, use and final disposal (ISO 14040, ISO 14041, ISO 14042, ISO 14043).

The functional unit was defined as 1 kg of packaged coffee delivered to the final consumer. The business has a wide product range but the functional unit was chosen in order to avoid allocation (in accordance with ISO 14041), with no distinction between the various products (e.g. different blends, different types of packaging, traditional and decaffeinated coffee, etc.).

System boundaries were defined to include all life cycle steps from coffee cultivation through to its distribution to consumers, consumption and disposal. Production of machinery and equipment are excluded from the system.

In figure 1 the coffee life cycle is presented: steps in the dotted boxes are not included in the study.

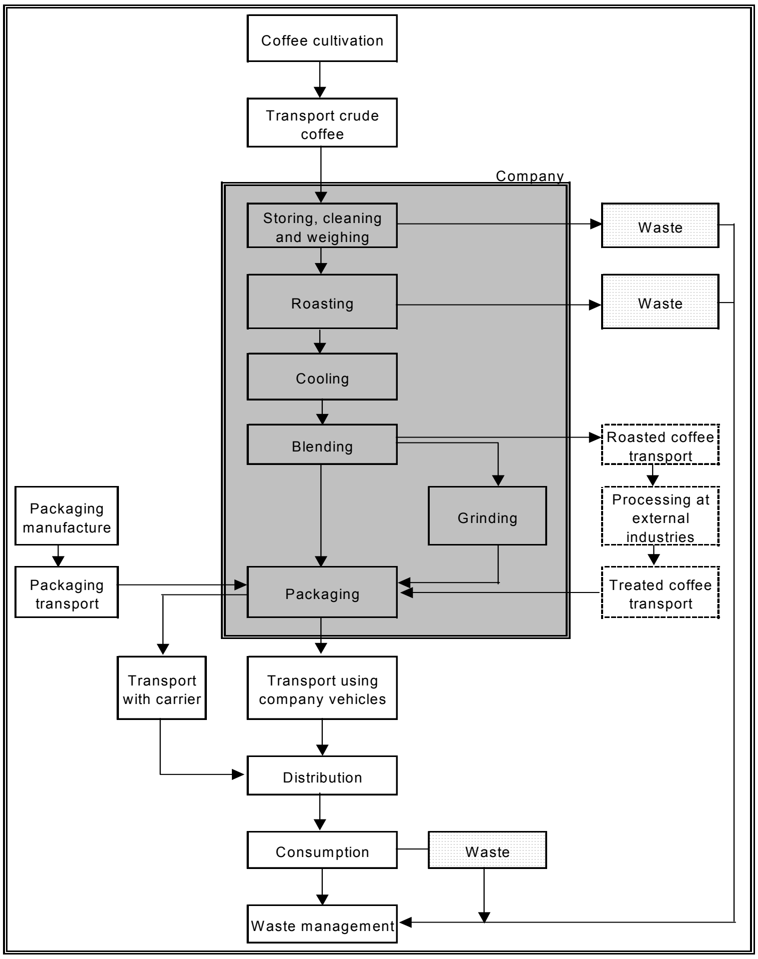

Figure 1. Coffee life cycle.

Goal and scope definition. The goal of the study is to examine the ways in which the coffee roasting and distribution company makes an impact on the environment in order to identify how to reduce its impacts and increase the environmental sustainability of the product from a life cycle perspective. It is important to ascertain the environmental aspects of coffee processing and include the environmental impacts connected with life cycle stages other than those relating solely to the company itself. The company wishes to use the information obtained from the LCA study as a starting point for the development of its environmental management set-up. This means collecting information about the entire life cycle of the product and using this information to improve the company's eco-efficiency.

Inventory analysis. The inventory analysis includes the following stages:
A) *Cultivation.* For this step literature data were collected, in particular for energy, fertilizer and pesticide use (Diers et al., 1999). Fertilizer and pesticide production data were included using commercially available databases (Ecobilan, 1999; Pré Consultants, 2003) while nitrogen and phosphorous emissions and pesticide emissions were quantified using estimation methods (Brentrup et al., 2000; Heathwaite, 2000; Hauschild, 2000). Average coffee production per hectare varies in relation to the type and characteristics of the land on which it is planted, together with other ecological factors, as well as to the age of plants. The approximate yield ranges from 2 to 6.5 quintals of finished product per hectare (Barbiroli, 1970). We assumed an average yield of 4.25 quintals/hectare.

Coffee beans can be processed in two ways: the dry method or the wet one. We assumed that only the dry method (also called the natural one) was used to process coffee beans and that the coffee cherries were both sun-dried and by using machines (assuming heavy fuel oil consumption of 0.11 l/kg) (Diers et al., 1999) and that the whole process was done by hand. This is very common in small or medium plantations and in regions where the temperatures are warmer and supplies of clean, fresh water are not plentiful. The dry method produces a single residue, the inner skin or outer hull, amounting to about 0.99t per 5.5t of coffee beans (Camaggio Sancinetti and Nicoletti, 1995).

B) *Processing.* For this step specific site data were collected for each basic process contained within the company box of the system flow chart. The direct material and energy inputs of the coffee processing and packing stage are: green coffee (or dried cherries); electricity (to power the equipment); natural gas (for the roasting step) and packing materials. The direct outputs are: roast coffee in primary and secondary packaging; air emissions (from natural gas combustion in the roaster) and waste (dust and scraps from cleaning and coffee chaff from roasting).

C) Packaging. The company uses many different types of primary and secondary packaging for roast coffee (aluminium cans, paper filters, etc). All of these have been included in the inventory analysis (specific site data) whilst commercially available databases have been used for the manufacturing of packaging materials (Ecobilan, 1999; Pré Consultants, 2003).

D) Transport. The main transportation activities take place at different life-cycle stages as follows:

1. Pesticides and fertilizers to coffee growers
2. Green coffee from growers to the coffee company premises;
3. Packaging from manufacturers to the coffee company premises;
4. Packaged coffee from the coffee company premises to local wholesalers and final points of sale;
5. Packaged coffee from the coffee company premises to national and international wholesalers;
6. Packaged coffee from each national and international wholesaler to final points of sale.

Point 1 is not included in the transport calculation. Primary data for points 2 and 3 were collected regarding distances travelled and quantities delivered. Primary data for point 4 were collected on diesel oil consumption for the quantities delivered (transported using company vehicles). In relation to point 5 it was extremely difficult to ascertain the deliveries made by carriers to each wholesaler. Therefore estimates of the average distance between the factory and a market town (discriminating between three market areas: regional, national and international) were made on the basis of data provided by the company. Point 6 is not included in the transport calculation since it was nearly impossible to collect accurate data about quantities delivered regionally, nationally and internationally to each supermarket and shop, and from these points of sale to each consumer. For these reasons this step is clearly underestimated.

E) Consumption. The consumption step is very difficult to measure and/or estimate because it depends on so many different factors: consumer nationality and tastes (the amounts of coffee and water used to make French coffee and Italian espresso differ greatly) or the type and brand of coffee machine used (in particular for energy consumption) amongst others and these differences are highly significant (+-30%). Nevertheless, in order to obtain some general information, selected data from a Pré Consultant LCA study (Pré Consultants, 2003) and specific information provided by Illycaffè Spa (Illycaffè Spa, 2003) were used. Data for the international market refer to two different filter coffee machines used by households throughout Europe (Pré Consultants, 2003): an electric aluminium coffee machine with a thermos jug (machine A) and a coffee maker for use on an ordinary gas stove (machine B); we assumed that 50% of the coffee delivered onto the international market was prepared with the first kind of machine and 50% with the second. We further assumed the use of 7-gram mono-dose filters. Data for the Italian market refer to an electric espresso coffee machine (machine C) used by households throughout Italy (Illycaffè Spa, 2003) assuming that 7 grams of coffee are used for a single cup of espresso.

The use of professional coffee machines is not included because they are far more complicated (they generally have other accessories that consume more energy). Water consumption (for coffee preparation and for cleaning the machine) and sugar are also excluded as they are assumed to be of little importance to the whole life cycle of the product and are also too difficult to model.

F) Disposal. Waste management includes packaging, coffee chaff and coffee grounds. We assumed that all these materials were disposed of without any recycling.

Data quality and assumptions are expressed in the previous description of the stages included in the inventory analysis. In general, specific on-site data were collected for the most important aspects of the life cycle; or were obtained from scientific literature and/or commercially available databases where on-site data were not available. The reference period for data collection was the year 2001. The LCA software used was TEAM 3.0 by Ecobilan (Ecobilan, 1999(*)).

Impact assessment: main results. The impact assessment step was performed investigating eight different impact categories (Table 1); Ecopoints were used as a general weighting factor. An ecopoint is a measure of the overall environmental impact of a particular product or process covering various environmental impacts (climate change, fossil fuel depletion, ozone depletion, human toxicity, waste disposal, acid deposition, eutrophication, etc.) obtained by adding together the score for each issue, calculated by multiplying the normalised impact with its percentage weighting (Braunschweig and Müller-Wenk, 1993, Baldo, 2000; Ecobilan 1999(*)).

Figure 2 shows the individual contributions of the process stages (in %) to the category results where the total of all contributions to each impact category is set at 100%.

Table 1. Impact categories.

Impact categories	Method	Unit
Air acidification	University of Leiden, Centre of Environmental Science (CML)	g eq. hydrogen (H^+)
Aquatic Eco-toxicity	University of Leiden, Centre of Environmental Science (CML)	$1e^3$ m^3
Eutrophication (water)	University of Leiden, Centre of Environmental Science (CML)	g eq. phosphates (PO_4^{3-})
Human toxicity	University of Leiden, Centre of Environmental Science (CML)	g
Terrestrial Eco-toxicity	University of Leiden, Centre of Environmental Science (CML)	t
Greenhouse effect (direct, 100 y.)	Intergovernmental Panel on Climate (IPPC)	g eq. carbon dioxide (CO_2)
Depletion of ozone layer	World Meteorological Organization (WMO)	g eq. trichlorofluoromethane (CFC-11)
Photochemical oxidant formation.	World Meteorological Organization (WMO)	g eq. ethylene

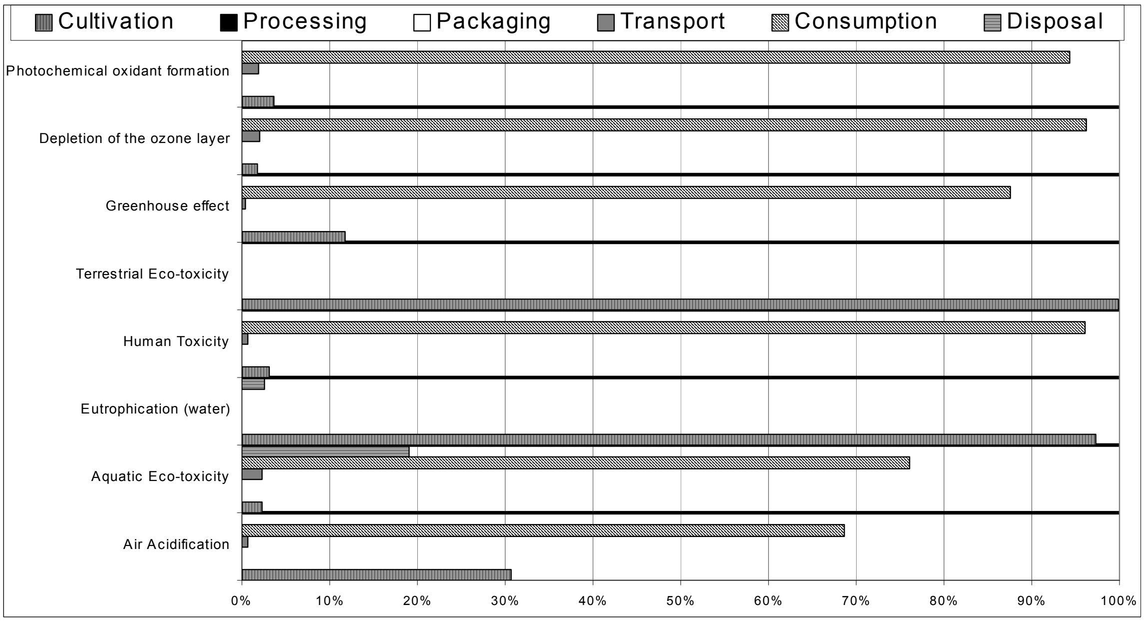

Figure 2. Impact categories studied.

From the figure it can be seen that the cultivation and the consumption stages make the greatest impacts. The cultivation stage contributes the most to Terrestrial Eco-toxicity and Eutrophication (contributions greater than 97%); the consumption stage contributes the most to Air acidification, Aquatic Eco-toxicity, Human Toxicity, Greenhouse effect, Depletion of ozone layer and Photochemical oxidant formation (contribution exceeds 68% for all categories cited). The disposal stage contributes to Aquatic Eco-toxicity (after consumption) and to Eutrophication (after cultivation). The contributions made by Transport are very limited but influence Photochemical oxidant formation, Greenhouse effect, Human Toxicity and Air acidification (after consumption and cultivation) and the Depletion of ozone layer and Aquatic Eco-toxicity (after consumption but before cultivation). The contributions of the processing and packaging stages are almost negligible (less than 1.7% for all categories).

The impact categories affected by process steps and the main emissions contributing thereto are set out in Table 2. Only process stages with a contribution higher than 5% to each impact categories are included in the table.

Table 2. Impact categories: main causes.

Process	Impact categories	%*	Mainly caused by	%*
Cultivation	Terrestrial Eco-toxicity	100	Copper	100
	Eutrophication (water)	97	Phosphates	97
	Air acidification	31	Ammonia (NH_3)	28
	Greenhouse effect	12	Carbon Dioxide (CO_2)	9
Consumption	Human Toxicity	96	Sulphur Oxides (SO_x)	60
	Depletion ozone layer	96	Halon 1301	96
	Photochemical oxidant for.	94	Hydrocarbons (except CH_4)	64
	Greenhouse effect	88	Carbon Dioxide (CO_2)	74
	Aquatic Eco-toxicity	76	Cadmium	43
	Air acidification	69	Sulphur Oxides (SO_x)	61
Disposal	Aquatic Eco-toxicity	19	Cadmium	12

* percentage referred to the total result of a given impact category

Figure 3 shows a general weighting factor based on Ecopoints (Braunschweig and Müller-Wenk, 1993, Baldo, 2000; Ecobilan 1999$_{(*)}$) relating to energy and waste, air emissions and water emissions for each life cycle stage considered. From the analysis of the figure it is evident that, in a general comparison among these three categories of ecopoints, air emissions are the most relevant and they are mainly caused by SO_2, NO_x, CO_2 emission in the consumption stage connected to energy consumption during the use of coffee machines.

Conclusions: interpretation of main results

For a better understanding of the importance of environmental management conducted under a life cycle perspective, figure 4 shows the Ecopoints applicable solely to the company (i.e. from figure 1 the steps included in the company box plus the step relating to distribution made by company vehicles). An analysis of environmental impacts made at company level alone in order to make environmental improvements would steer management towards targeting almost exclusively the distribution stage (e.g. improvements to the company vehicle pool) and the coffee roasting stage (e.g. improving energy consumption, air emissions and waste management).

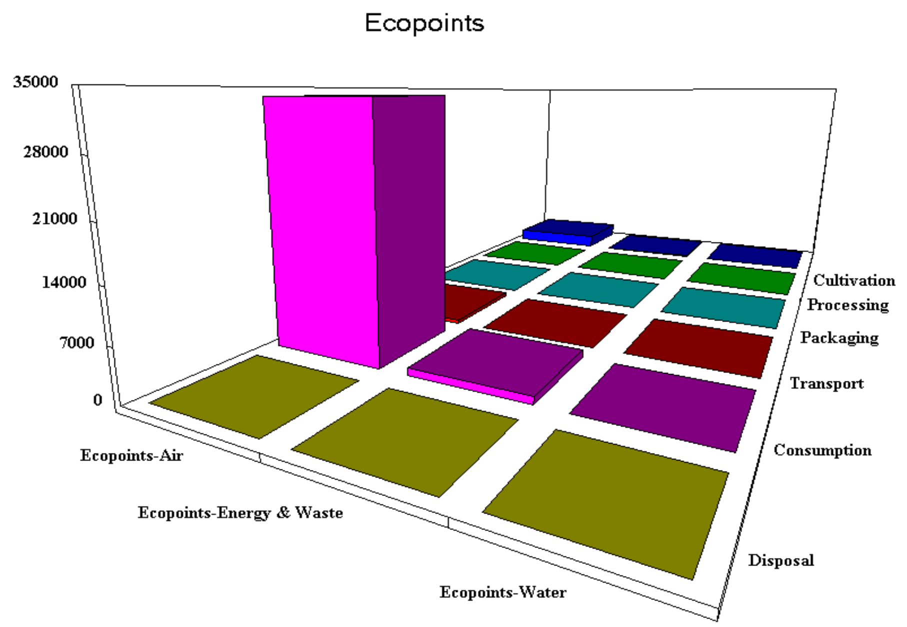

Figure 3. Ecopoints.

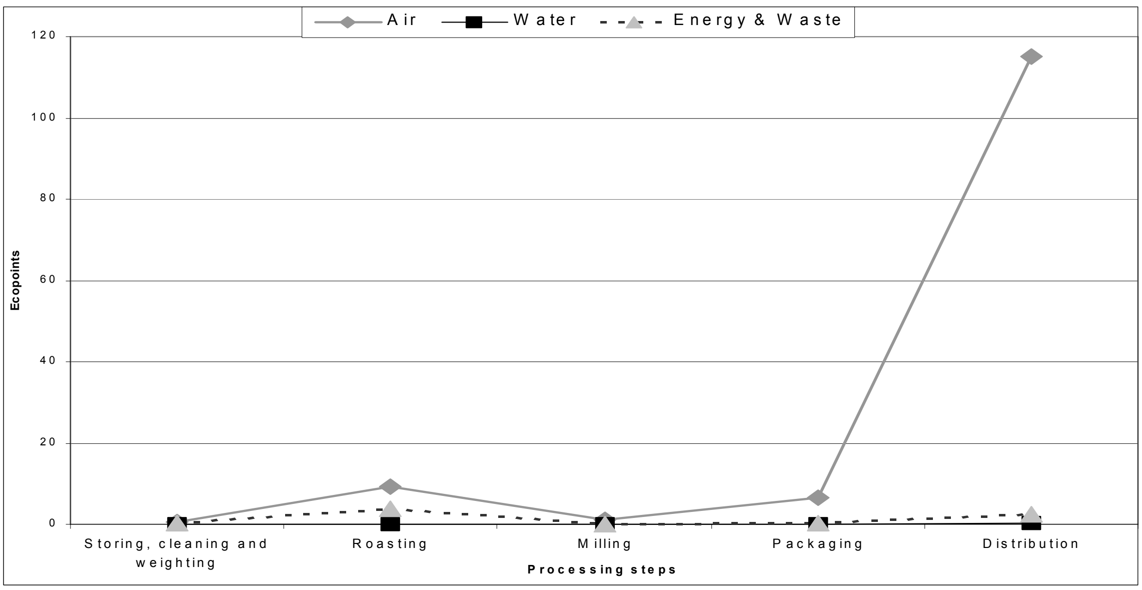

Figure 4. Company level: Ecopoints.

Accordingly, at company level the main environmental improvements that could be addressed are:

- *Air emissions* – principally due to fuel consumption of vehicles for local deliveries (all the vehicles used for local deliveries run on diesel fuel) and to a lesser extent from the combustion of fossil fuels in the roaster (natural gas). Improvements in fuel consumption would enable air emissions to be lowered .

- *Energy consumption* – electricity consumption refers to single processing steps and to fork-lifts that are powered by electric batteries. Also improvements in energy efficiency would enable air emissions to be lowered.

- *Waste management* – waste management at company level is mainly related to coffee chaff. At present this solid waste is disposed of alongside other urban refuse. Although the company would be interested in seeking an alternative use, it has been discouraged from taking any initiative due to the small quantities concerned.

Under a life cycle perspective the company should approach its environmental management decision making differently and concentrate mostly on the cultivation and consumption stages.

- *Cultivation data* – although not all the data relating to cultivation have been included, it is evident (see figure 2) that this step has a significant impact on the entire coffee life cycle, therefore it is fundamental for the company to include the data in its environmental considerations. Environmental improvements could most probably be achieved by choosing organic and/or sustainable coffee farms as suppliers.

- *Solid wastes* – at company level the main solid waste is coffee chaff, but when the consumption step is also taken into consideration then coffee grounds make up the largest proportion of solid waste (apart from packing materials). Instead of being disposed of, coffee grounds could be used as food for worms as well as for compost. The company should place bins for composting food waste in each point of consumption to which they deliver. These can then be collected when subsequent deliveries are made and contents used for composting in a worm farm. The worms would process the coffee grounds into fertilizers. The amounts obtained through collecting coffee grounds in this way together with the coffee chaff produced on the company premises could allow it to start up a small-scale enterprise for vermiculture or compost production.

- *Energy consumption* – other methods to increase the environmental sustainability of the company could involve setting up projects in collaboration with manufacturers of coffee machines where such projects undertake joint research aimed at improving energy efficiency. Energy waste could also be prevented by information campaigns to heighten consumer awareness (both at professional and household level).

- *Packaging* – even though this step makes no great impact, investigating more recyclable alternatives to the current types of packaging used could nevertheless be very worthwhile.

From the above LCA clearly emerges as a useful tool to provide information for effective environmental management under a life cycle perspective, and as one that does not limit improvement opportunities to the physical confines of a company alone.

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References

Baldo G.L., *LCA: Life Cycle Assessment,* Ipa servizi Editore, Milano, Italia 2000;

Barbiroli G. *Produzione e commercio internazionale del caffè* Riccardo Patron Bologna 1970;

Braunschweig A. et Müller-Wenk R., *Oekobilanzen für Unternehmungen. Eine Wegleitung für die PraxisVerlag,* Paul Haupt, Bern, pp.224. 1993. Updated version by BUWAL, 1998;

Brentrup F., Küsters J., Lammel J., Kuhlmann H. *Methods to estimate on-field nitrogen emissions from crop production as an input to LCA studies in the agricultural sector* International Journal of LCA 5 (6) 200 pg. 349-357;

Camaggio Sancinetti G., Nicoletti G.M. *Il ciclo di lavorazione del caffè ed i sui sottoprodotti Nota 1: Aspetti quantitativi* Industrie alimentari XXXIV 1995 pg. 1137-1146;

Diers A., Langowski H.-C., Pannkoke K., Hop R., *Produkt-Ökobilanz vakuumverpackter Röstkaffee* Ecomed publishers LCA Documents vol.3 1999;

Ecobilan Group *Data for Environmental Analysis and Management* (DEAM™) Version 3.0, 1999;

Ecobilan Group *Tools for Environmental Analysis and Management* (TEAM™) Version 3.0, 1999(*);

Hauschild M., *Estimating pesticide emissions for LCA of agricultural products* in Weidema B.P., Meeusen M.J.G. *Agricultural data for Life Cycle Assessment* Agricultural Economics Research Institute (LEI), The Hague, 2000, Vol. II;

Heathwaite L., Flows of phosphourous in the environment: identifying pathways of loss from agricultural land in Weidema B.P., Meeusen M.J.G. Agricultural data for Life Cycle Assessment Agricultural Economics Research Institute (LEI), The Hague, 2000, Vol. II;

Illycaffè Spa e.mail 4/4/03;

ISO 14040: 1997 Environmental management. Life cycle assessment. Principles and framework;

ISO 14041: 1998 Environmental management. Life cycle assessment. Goal and scope definition and inventory analysis;

ISO 14042: 2000 Environmental management. Life cycle assessment. Life cycle impact assessment;

ISO 14043: 2000 Environmental management. Life cycle assessment. Life cycle interpretation;

Mirulla R., La gestione ambientale nella supply chain: esperienze e metodi di coinvolgimento dei fornitori in Scaramazza R. La fabbrica verde. Certificazione ambientale e imprese sostenibili, Nuovo Studio Tecna, Roma 2001;

Pré Consultants SimaPro 5.1, Amersfoort, Netherlands, 2003; www.wri.org.

Slaughter offal to feed or to fuel - a comparison of systems in terms of energy balances, land use and emissions

Hans B. Wittgren[1*], *Helena Elmquist*[2] *and Ingrid Strid Eriksson*[2]

1. JTI - Swedish Institute of Agricultural and Environmental Engineering, P.O. Box 7033, SE-750 07 Uppsala, Sweden

2. SLU - Swedish University of Agricultural Sciences, Department of Biometry and Engineering, P.O. Box 7032, SE-750 07 Uppsala, Sweden

** Corresponding author. Current address: Linköping University, Environmental Science Programme, Campus Norrköping, SE-601 74 Norrköping, Sweden, Phone: +46 11 36 33 48, E-mail: hb.wittgren@ituf.liu.se*

Abstract

Large quantities of slaughter offal are being processed to meat- and bonemeal, but due to the current legislation its use as animal feed is very restricted. One option is to use slaughter offal directly as fuel in incineration plants. This study compared two alternative systems: (1) Processing of slaughter offal to meat- and bonemeal (feed) and to animal fat (fuel), and (2) Direct incineration of slaughter offal (fuel). In order to make the two alternatives comparable, the core systems were expanded with complementary systems to produce the same amounts of feed, generate the same amounts of energy and use the same areas of arable land. The analysis of the two alternatives used existing life cycle data and assessments in combination with the two simulation tools ORWARE (ORganic WAste REsearch model) and SALSA (Systems AnaLysis for Sustainable Agriculture). The comparison showed that processing of slaughter offal to feed (alt. 1) was superior to using it as fuel (alt. 2) in terms of both energy balances and all considered emission impact categories (global warming, acidification and eutrophication potential). The difference between the two alternatives may be interpreted as the price, in terms of energy and emissions, that has to be paid to lower the risk for recycling of the bovine spongiform encephalopathy (BSE) agent.

Keywords: energy balances, environmental systems analysis (ESA), life cycle assessment (LCA), slaughter offal, substance flow accounting (SFA).

Introduction

Following the discovery of bovine spongiform encephalopathy (BSE) in 1986, the countries of Europe and the European Union have successively implemented several measures to prevent recycling of the BSE agent (Heim and Kihm, 2003). Some of the most important measures have implied banning of feeding meat- and bonemeal (MBM) to farm animals. Initially, only feeding of MBM to ruminants was banned (European Commission, 1994), but due to the risk for cross-contamination most countries in Europe banned feeding of MBM to all farm animals in 2001 (European Commission, 2000). Although it is possible that this total feed ban will be relaxed in the future, it is necessary to consider alternative options for handling of slaughter offal.

The most obvious alternative is to use the slaughter offal as fuel by direct incineration in power plants. This should be a safe option in terms of managing BSE, but how does direct incineration come out in comparison with production of MBM as feed in terms of resource use and environmental impact? Such a comparison should provide part of the background for decisions regarding the future handling of slaughter offal. In addition, analysis of different systems for handling slaughter offal should provide necessary input to life cycle assessments (LCAs) of meat production.

The objective of this study was to make a quantitative comparison of two alternative systems (Figures 1 and 2):

1. Processing of slaughter offal to MBM (feed) and to animal fat (fuel)
2. Direct incineration of slaughter offal (fuel)

The comparison was based on systems analysis in terms of energy balances, land use and emissions.

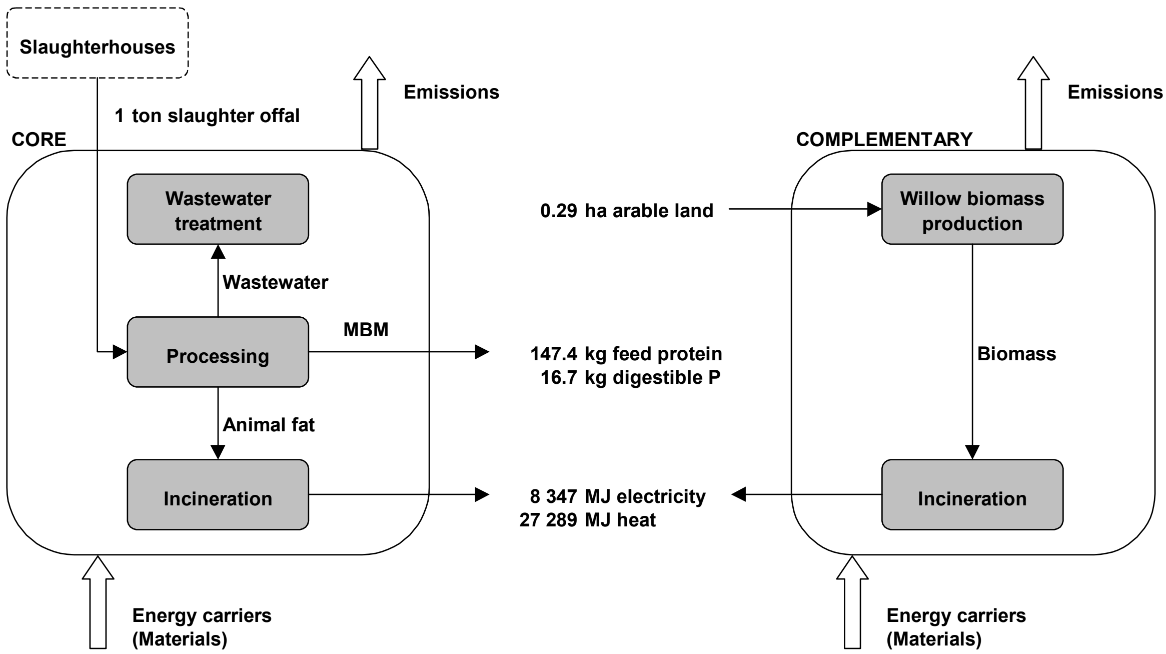


Figure 1. Scheme of alternative 1: Processing of slaughter offal to MBM (feed) and to animal fat (fuel).

Methods

Functional unit, core and complementary systems

The two alternatives were designed to treat ‘1 ton (wet weight) of slaughter offal’ - the functional unit (FU) of the analysis. This treatment was performed in the ‘core’ systems (Figures 1 and 2). To make the two alternatives comparable, also ‘complementary’ systems were needed (Figures 1 and 2). These were designed according to the following logic:

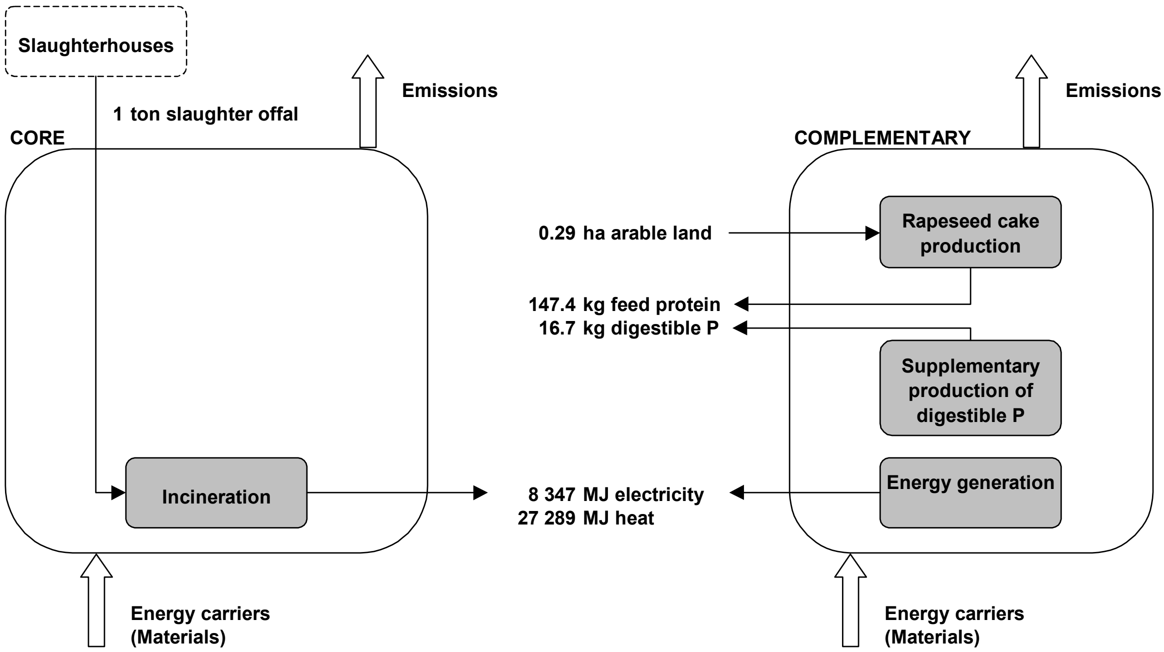


Figure 2. Scheme of alternative 2: Direct incineration of slaughter offal (fuel).

The core system of alt. 1 produced MBM containing certain amounts of feed protein and digestible phosphorus (P). Corresponding amounts were generated in alt. 2 by including feed production, based on rapeseed and inorganic P, in the complementary system. The production of rapeseed required arable land, and an equivalent area of arable land was used in the complementary system of alt. 1 for production of willow biomass. This choice was motivated by the fact that it is desirable to increase the production of renewable energy carriers. The willow biomass was incinerated to produce electricity and heat. Finally, energy generation in a natural gas powered plant was required in the complementary system of alt. 2, to balance the difference between energy generated from incineration of animal fat + willow biomass, in alt. 1, and incineration of slaughter offal, in alt. 2. With this design, both alternatives produced the same amounts of feed protein and digestible P, used equivalent areas of arable land and generated the same amounts of electricity and heat (Figures 1 and 2). The feed protein, however, was not of the same quality in the two alternatives, since the amino acid composition of MBM and rapeseed cake are quite different.

In terms of amino acid composition, soybean meal is more similar to MBM than rapeseed cake (Rodehutscord et al., 2002). Therefore we analysed also feed production based on soybean meal (from Brazil), and compared it with feed production based on rapeseed cake (from Sweden). A complete system with soybean meal could, however, not be analysed, since we did not have data to set up a realistic land use alternative to soybean (such as willow is in the case of rapeseed).

Material flows, such as phosphorus ore or process chemicals, were not included in the analysis. Furthermore, the building phases of processing, wastewater treatment, and power plants were not considered.

Sub-systems

In the following, the different sub-systems (grey boxes in Figures 1 and 2) are briefly described in terms of data sources, assumptions and calculations. Also the soybean meal subsystem, and how transportation was handled, is described.

Processing (in alt. 1): In Sweden only one company, Konvex AB, processes slaughter offal to MBM and animal fat. The data used in this study were obtained from their largest plant, Krutmöllan, which processes approximately 60,000 tons (wet weight) annually (Konvex AB, 2001; L. Virta, personal communication, January 15, 2003). The composition of incoming slaughter offal and the resulting products are summarised in Table 1. Data on energy use and emissions, for the Processing sub-system as well as for the others, were recalculated to relate to the FU (Tables 2 and 3 in the Results section).

Table 1. Composition of slaughter offal and the products meat- and bonemeal (MBM) and animal fat at the Konvex plant in Krutmöllan, Sweden (expressed as kg per ton wet weight of slaughter offal, e.g., the functional unit (FU)).

Component	Water	Dry matter	Protein	Fat	Digestible P [2]
Fraction			*kg FU^{-1}*		
Slaughter offal [1]	650	350	147	150	16
Meat- and bonemeal (MBM)	7	220	147	20	16
Animal fat	3	130	0	130	0

1. No data was available on the content of protein, fat and phosphorus in the slaughter offal. These values have been calculated from the data on product contents, assuming no losses in the process. According to Konvex AB, losses are < 1% (L. Virta, personal communication, January 15, 2003)
2. The content of digestible P in MBM was calculated from data on total P content, assuming that 81% is digestible (Rodehutscord et al., 2002)

Wastewater treatment (in alt. 1): The wastewater was first treated at the slaughter offal processing plant, and this treatment was included in the Processing sub-system (above) in terms of energy use and emissions. The pre-treated wastewater was piped to the municipal wastewater treatment plant in Kävlinge for further treatment. Data from the wastewater treatment plant showed that the share coming from the slaughter offal processing plant made up for 3.1% (flow), 3.7% (BOD7), 18.8% (total nitrogen) and 2.4% (total phosphorus) of the total load in 2001 (Kävlinge kommun, 2002). Based on these shares, and somewhat arbitrary, 10 % of the energy use in the wastewater treatment plant was allocated to the treatment of wastewater from the slaughter offal processing plant.

Incineration (in alt. 1 and 2): The Swedish ORWARE (ORganic WAste REsearch) model has a sub-model for incineration of organic waste in a combined heat and power plant (Björklund, 1998; Eriksson et al., 2002). This sub-model was used to calculate energy balances and emissions for incineration of animal fat and willow biomass in alt. 1, and for incineration of slaughter offal in alt. 2.

Willow biomass production (in alt. 1): An annual average biomass yield of 6400 kg (dry weight) ha^{-1} was calculated from harvest estimates presented by the Swedish company Agrobränsle AB (2003). Energy use and emissions (expressed as global warming potential, acidification potential and eutrophication potential) in a willow biomass cropping system were obtained from an LCA performed for conditions in New York state by Heller et al. (2003). Their assessment considered an estimated 23 year lifespan of a cropping system with seven harvest rotations. The system included field preparation, planting, weed control, coppice, fertilisation, harvest, willow stool elimination, and production of various inputs such as herbicides and fertilisers. In our study we used their base case, where ammonium sulphate was used as fertiliser, and recalculated the results to relate them to the FU.

Rapeseed cake production (in alt. 2): One part (SALSA-arable) of the Swedish SALSA (Systems AnaLysis for Sustainable Agricultural production) model was used for calculations of energy use and emissions in rapeseed cake production (Elmquist et al., 2003). SALSA, similar to ORWARE (mentioned above), is an energy and substance flow accounting model (SFA) complemented with life cycle assessment (LCA) methodology for evaluation of environmental impacts (Elmquist et al., 2004; Strid Eriksson et al., 2004).

The rapeseed cake production system in SALSA-arable includes different field operations, the soil/crop system, drying and preparation of rapeseed oil and cake, and production of inputs such as seed and fertiliser. The yield of rapeseed was based on Swedish harvest data for winter and spring rapeseed from the dominating production regions (Skåne and Östergötland, respectively). It was further assumed that the rapeseed harvest was processed, with 73% exchange, to rapeseed cake, containing 32% protein and 1.3% P. Digestible P was assumed to represent 30% of the P content (Rodehutscord et al., 2002).

These assumptions resulted in a protein (in rapeseed cake) harvest of 501 kg ha^{-1} yr^{-1} and a need for 0,29 ha to equal the production of MBM protein per FU in alt. 1. Based on market prices, the value of rapeseed cake was assumed to be 33% of the total value of the rapeseed yield. This share of energy use and emissions for production of rapeseed was allocated to rapeseed cake, and the rest to rapeseed oil. Since the inclusion of rapeseed cake in the comparison was managed via an allocation procedure, there was no need for a sub-model analysing the fate of the rapeseed oil.

The amount of digestible P in rapeseed cake was only 1.8 kg FU^{-1} and the feed had to be complemented with inorganic P to contain the same amount as in alt. 1. Data on energy use and emissions for production of P fertiliser were used (Sundqvist et al., 2002), along with the estimate that 80% of the P content is digestible (Rodehutscord et al., 2002).

Soybean meal production (for comparison with rapeseed cake production): The SALSA-soybean (Strid Eriksson et al., 2004) description of the production system for soybean meal (of Brazilian origin) include different field operations, the soil/crop system, drying of seeds, transport from farm to extraction plant, extraction, transport from extraction plant to harbour, ocean transport from Brazil to Germany, sea transport from Germany to Sweden, and production of inputs such as seed and fertiliser. A soybean harvest of 2200 kg ha^{-1} yr^{-1} was assumed, with a 70% exchange of soybean meal containing 40% protein.

These assumptions resulted in a protein (in soybean meal) harvest of 616 kg ha^{-1} yr^{-1} and a need for 0,24 ha to equal the production of MBM protein per FU in alt. 1. The value of soybean meal was assumed to be 70% of the total value of soybean, and energy use and emissions allocated accordingly. The digestible P deficit was corrected for in the same way as for rapeseed cake.

Energy generation (in alt. 2): It was assumed that the difference between electricity and heat generated by incineration of animal fat + willow biomass (in alt. 1) and incineration of MBM (in alt. 2) was generated in a heat and power plant fuelled with natural gas. Data on conversion efficiency and emissions from a plant in Sweden (Heleneholmsverken) were obtained from Uppenberg et al. (2001). The overall energy conversion efficiency was 94%, out of which 25% was electricity and 75% heat.

Transportation: Transportation of the different products were not included in the analysis, with the exeption of the transports concerning soybean meal production (mentioned above). However, separate calculations of energy use and emissions were made for the transportation of slaughter offal to processing and willow biomass to incineration. Data on transport distances and amounts concerning slaughter offal to processing were obtained from the processing company (Konvex AB). For willow biomass, a transportation distance of 100 km between biomass production and incineration was assumed. Data on diesel consumption and emissions for a heavy truck with trailer (load capacity: 35 tons) were taken from Sundqvist et al. (2002).

Impact assessment
Calculations of energy use included also upstream use, for production of the energy carriers, to yield primary energy use (Eriksson et al., 2002; Sundqvist et al., 2002). All results on energy use are thus expressed as primary energy. Upstream emissions from production of energy carriers were also calculated and are included in results on emissions.

Emissions to air and water were recalculated to the impact categories global warming potential (time horizon 100 years), acidification potential and eutrophication potential using weighting factors according to Houghton et al. (1996) - for global warming potential - and according to Lindfors et al. (1995) - for acidification (maximum scenario) and eutrophication potential.

Results

The primary energy use was 4.3 times larger in alt. 2 as compared to alt. 1, mainly due to the generation of electricity and heat from natural gas needed to compensate for the incineration of willow biomass (Table 2).

The sum of primary energy use in soybean meal production was 2458 MJ FU^{-1} (not shown) as compared to 1012 MJ FU^{-1} for rapeseed cake production. The difference was mainly due to oil consumed for ocean transportation.

Table 2. Primary energy use per functional unit (FU) in different sub-systems of the two alternatives (Figures 1 and 2).

	Primary energy use					
	Electricity	Diesel	Natural gas	Oil / Coal	Unspecified [1]	*Sum*
	MJ FU^{-1}					
Alternative 1						
Processing of slaughter offal	963		3 232			*4 195*
Incineration of animal fat	72					*72*
Wastewater treatment	27			2		*29*
Willow biomass production		578			679	*1 257*
Incineration of willow biomass	2 052					*2 052*
Sum	*3 115*	*578*	*3 232*	*2*	*679*	***7 606***
Alternative 2						
Incineration of slaughter offal	615					*615*
Rapeseed cake production	142	256	559	55		*1 012*
Supplementary production of digestible P	153		161	482		*795*
Energy generation			30 440			*30 440*
Sum	*910*	*256*	*31 160*	*536*	*0*	***32 862***

1 The source of data on willow biomass production (Heller et al., 2003) did not specify energy carriers beside diesel.

The global warming potential (Table 3) reflected the energy use to a large extent, since CO_2-emissions from incineration and energy generation were the dominant contributions. Emissions of N_2O were of the same magnitude in the two alternatives, 185 and 231 kg CO_2-equivalents FU^{-1}, and thus a substantial contribution, 37%, in alt. 1, compared to only 9% in alt. 2. Willow biomass and rapeseed cake production were the most important sources of N_2O in alt. 1 and 2, respectively.

The sum of emissions contributing to acidification potential, particularly NO_x and to some extent SO_2, were similar in alt. 1 and alt. 2 (Table 3). It was, however, noted that the acidification potential was a factor 3.4 larger for willow biomass production than for rapeseed cake production, and a factor 5.6 larger for soybean meal production than for rapeseed cake production (not shown). In the later case, larger energy use and less efficient NO_x reduction during combustion in the soybean meal production were the reasons. Concerning willow biomass production, it was assumed that the larger emissions, in comparison with rapeseed cake production, were at least partly a consequence of the different sources of data for calculations of emissions from, e.g., field operations and fertiliser manufacturing.

Eutrophication potential was more than twice as a high in alt. 2 as compared to alt. 1 (Table 3). The difference was a result of very different assumptions regarding leaching of nitrate and phosphorus in willow biomass production and rapeseed cake production, respectively. The assessment of willow biomass production assumed no leaching (Heller et al., 2003), while the SALSA-arable model assumed leaching of 94 kg NO_3^- ha^{-1} yr^{-1} and 0.9 kg P ha^{-1} yr^{-1} from rapeseed production. SALSA-soybean assumed similar leaching from soybean production, but since 70% was allocated to soybean meal, as compared to only 33% to rapeseed cake, the leaching per FU was higher in the case of soybean meal. In addition, NO_x-emissions were higher for soybean meal production, making the total eutrophication potential 111 kg O_2-consumption FU^{-1} (not shown) as compared to 55.6 kg in rapeseed cake production (Table 3).

In alt. 1, transportation of slaughter offal to processing and of willow biomass to incineration used 509 and 327 MJ FU^{-1}, respectively. The reason for higher use in transportation of slaughter offal, although quantities were smaller, is that average distances are much longer than was assumed for willow biomass transportation (100 km). Taken together, these two transports contributed 63 kg CO_2-equivalents FU^{-1} to global warming potential, 0.5 kg SO_2-equivalents FU^{-1} to acidification potential, and 3.0 kg O_2-consumption FU^{-1} to eutrophication potential. If these were the only transports, they would increase primary energy use in alt. 1 by 11%, global warming potential by 13%, acidification potential by 9% and eutrophication potential by 10%.

Discussion

The comparison showed that processing of slaughter offal to feed (alt. 1) was superior to using it as fuel (alt. 2) in terms of both energy balances and all three emission impact categories. The difference between the two alternatives may be interpreted as the price, in terms of energy and emissions, that has to be paid to lower the risk for recycling of the BSE agent.

The comparison did not take into account the fact that protein from MBM and rapeseed cake have different amino acid composition. It is therefore not realistic to fully exchange one for the other in feed preparation (Rodehutscord et al., 2002). In a more stringent comparison, alt. 2 would thus have to be complemented with further sub-systems for production of supplementary feed ingredients, probably leading to an even larger difference in terms of energy use and emissions.

Table 3. Emissions to air and water expressed as the three impact categories global warming (kg CO_2-equivalents), acidification (kg SO_2-equivalents) and eutrophication (kg O_2-consumption) per functional unit (FU) for different sub-systems of the two alternatives (Figures 1 and 2).

	Global warming *kg CO_2-eq. FU^{-1}*	Acidification *kg SO_2-eq. FU^{-1}*	Eutrophication *kg O_2-cnsmp. FU^{-1}*
Alternative 1			
Processing of slaughter offal	211	1.4	5.7
Incineration of animal fat	1	0.3	2.2
Wastewater treatment	0	0.0	1.7
Willow biomass production	228	1.6	7.6
Incineration of willow biomass	59	1.9	14.6
Sum	***499***	***5.2***	***31.8***
Alternative 2			
Incineration of slaughter offal	69	0.4	3.6
Rapeseed cake production	238	0.5	55.6
Supplementary production of digestible P	54	0.6	1.1
Energy generation	2 091	4.2	30.8
Sum	***2 451***	***5.7***	***91.1***

Soybean meal is more similar to MBM and its use as a feed ingredient has increased following the ban on feeding of MBM to farm animals. In comparison with rapeseed cake production, soybean meal production was calculated to have higher energy use and larger emissions. The major reason is that soybean meal was assumed to be produced in Brazil, with ocean transportation to Sweden included in the calculations, while rapeseed cake was assumed to be produced locally in Sweden, with no transportation needed.

It is likely that road transportation (not included in Tables 2 and 3) would add more to energy use and emissions in alt. 1 than in alt. 2. Transport distances for slaughter offal would probably be shorter for alt. 2 since it is reasonable to assume that incineration can be implemented at several places, while in alt. 1 all slaughter offal is taken to one single processing plant. A further reason is that the weight of willow biomass (3.8 tons FU^{-1}) is much larger than the weights of any of the other products involved in the comparison.

Since energy balances were one focus of the study, willow biomass production was chosen as alternative to rapeseed production concerning land use. Another option could have been to consider permanent fallow as the alternative land use. This would have made the comparison turn around since complementary energy generation would have been needed in alt. 1 instead of alt. 2. Permanent fallow was, however, considered as too different economically from rapeseed production, when the farm economics dimension was not included in the comparison. This is not to say that alt. 1 and alt. 2 in the present study were economically equal. To serve as true decision support, most comparisons of systems need to include economic and social dimensions in addition to the resource and environmental dimensions.

The present study took short-cuts in the sense that already existing tools and assessments, not designed to analyse systems for management of slaughter offal, where used. This is sometimes necessary, since the choice is between a simplified analysis or no analysis at all. The alternative of a seam-less assessment, with tailor-made tools, is presently not economically possible in most applications. This state-of- the-art points to the need for development of agri-food sector assessment tools that are highly compatible with each other.

References

Agrobränsle AB (2003). *Kalkyl* [online]. Agrobränsle AB, Örebro, Sweden. Available from: http://www.agrobransle.se [Accessed 7 July 2003]. (In Swedish).

Björklund, A., 1998. *Environmental systems analysis of waste management with emphasis on substance flows and environmental impact*. Thesis (licentiate). TRITA-KET-IM 1998:16, Department of Chemical Engineering and Technology, KTH - Royal Institute of Technology, Stockholm, Sweden.

Elmquist, H., Strid Eriksson, I., Öborn, I. and Nybrant, T., 2003. An environmental systems analysis of winter wheat, barley and rapeseed production on an arable farm in Sweden - A study on effects of N fertilisation rates. *Journal of Cleaner Production*, submitted manuscript.

Elmquist, H., Lindgren, U. and Mäkilä, K., 2004. *Decision-making and environmental impacts - A dynamic simulation model of a farm business*. Food 21 Report, Department of Biometry and Engineering, SLU - Swedish University of Agricultural Sciences, Uppsala, Sweden, in press.

Eriksson, O., Frostell, B., Björklund, A., Assefa, G., Sundqvist, J.-O., Granath, J., Carlsson, M., Baky, A., and Thyselius, L., 2002. ORWARE - A simulation tool for waste management. *Resources, Conservation and Recycling* **36** (4), 287-307.

European Commission (EC), 1994. Commission Decision 94/381/EC of 27 June 1994 concerning certain protection measures with regard to bovine spongiform encephalopathy and the feeding of mammalian derived protein (Text with EEA relevance). *Off. J. Eur. Communities* **L 172** (07/07/1994), 23-24.

European Commission (EC), 2000. Council Decision 2000/766/EC of 4 December 2000 concerning certain protection measures with regard to transmissible spongiform encephalopathies and the feeding of animal protein. *Off. J. Eur. Communities* **L 306** (07/12/2000), 32-33.

Heim, D. and Kihm, U., 2003. Risk management of transmissible spongiform encephalopathies in Europe. *Rev. sci. tech. Off. int. Epiz.* **22** (1), 179-199.

Heller, M. C., Keoleian, G. A. and Volk, T. A., 2003. Life cycle assessment of a willow bioenergy cropping system. *Biomass and Bioenergy* **25**, 147-165.

Houghton, J. T., Meiro Filho, L. G., Callander, B. A., Harris, N., Kattenburg, A. and Maskell, K. (eds), 1996. *Climate Change 1995: The Science of Climate Change: Contribution of Working Group I to the Second Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge University Press, Cambridge, UK.

Konvex AB, 2001. *Konvex AB Krutmöllan - Ansökan om tillstånd enligt 9 kap. miljöbalken*. Konvex AB, Lidköping, Sweden. (In Swedish).

Kävlinge kommun, 2002. *Kävlinge avloppsreningsverk år 2001 - Miljörapport enligt miljöskyddslagen*. Kävlinge kommun, Kävlinge, Sweden. (In Swedish).

Lindfors, L.-G., Christiansen, K., Hoffman, L., Virtanen, Y., Juntilla, V., Hanssen, O.-J., Rønning, A., Ekvall, T. and Finnveden, G., 1995. *Nordic Guidelines on Life-Cycle Assessment*. Nord 1995:20, Nordic Council of Ministers, Copenhagen, Denmark.

Rodehutscord, M., Abel, H. J., Friedt, W., Wenk, C., Flachowsky, G., Ahlgrimm, H.-J., Johnke, B., Kühl, R. and Breves, G., 2002. Consequences of the ban of by-products from terrestrial animals in livestock feeding in Germany and the European Union: alternatives, nutrient and energy cycles, plant production, and economic aspects. *Arch. Anim. Nutr.* **56**, 67-91.

Strid Eriksson, I., Elmquist, H., Stern, S., Nybrant, T., 2004. Environmental systems analysis of pig production - the impact of feed choice. *Int. J. LCA.*, submitted manuscript.

Sundqvist, J.-O., Baky, A., Carlsson, M., Eriksson, O. and Granath, J., 2002. *Hur skall hushållsavfallet tas om hand? Utvärdering av olika behandlingsmetoder*. IVL Report B 1462, IVL - Swedish Environmental Research Institute, Stockholm, Sweden. (In Swedish with English summary).

Uppenberg, S., Brandel, M., Lindfors, L.-G., Marcus, H.-O., Wachtmeister, A., Zetterberg, L., 1999. *Miljöfaktabok för bränslen - Del 2. Bakgrundsinformation och teknisk bilaga*. IVL Report B 1334B-2, IVL - Swedish Environmental Research Institute, Stockholm, Sweden. (In Swedish).

Environmental assessment of contrasting pig farming systems in France

*Basset-Mens, C.** and van der Werf, H.M.G*

: INRA, UMR Sol Agronomie Spatialisation de Rennes-Quimper, 65, rue de Saint Brieuc CS 84215, 35 042 Rennes Cedex, France. Tel : +33 02 23 48 52 22. Fax : +33 02 23 48 54 30. Email : basset@roazhon.inra.fr

Abstract

The aim of this study was to produce a multicriteria environmental assessment of three contrasting pig farming systems : conventional Good Agricultural Practice (GAP), "Label Rouge" (LR) and "Agriculture Biologique" (AB), using the Life Cycle Assessment (LCA) method. Average, favourable and unfavourable scenarios were defined and evaluated for each production mode for seven impact categories. Expressed per hectare, LR and AB had lower impacts than GAP for eutrophication and acidification, a higher (LR) or similar (AB) impact for climate change and was less productive (respectively - 14% and – 45%). We identified "hot-spots" for each system. However, for the three scenarios important margins of improvement were highlighted.

Keywords: LCA; pig; uncertainty; hot-spots.

Introduction

In Brittany (France), the successful development of intensive pig production since Word War II has led to a severe degradation of the environment. Hence, the current pig production model is in crisis. Although alternative farming systems are favourably considered by society, none of these have yet been assessed for their environmental impact as far as pig production is concerned. The global diagnosis of the environmental performance of current and more prospective farming systems appears as an emerging need and this evaluation should include an estimation of the uncertainty of the main results (Huijbregts et al., 2001; Guinée et al., 2002). Among environmental assessment approaches, Life Cycle Assessment (LCA) has been identified as a valuable tool for the environmental evaluation of farming systems (van der Werf and Petit, 2002). The objectives of this study were to explore the diversity of pig farming systems using LCA with a prospective view and to assess the robustness of the results. We were facing the following methodological issues: (i). How should contrasting farming systems with different degrees of optimisation be described and how should an environmental inventory be carried out? (ii). How should the uncertainty of the results be assessed? This paper proposes a scenario-based approach for the environmental assessment of contrasting farming systems including an estimation of the uncertainty of the results.

Materials and methods

Evaluation methodology

Environmental impacts associated with pig farming systems were evaluated using Life Cycle Assessment (LCA). In this approach, the potential environmental impacts of a product are assessed by quantifying and evaluating the resources consumed and the emissions to the environment at all stages of its life cycle – from the extraction of resources, through the production of materials, product parts and the product itself, and the use of the product to its reuse, recycling or final disposal (Guinée et al., 2002). The present study only deals with the processes up to and including the production on the farm. We compared three scenarios for pig production. The Good Agricultural Practice (GAP) scenario corresponds to conventional production, optimised in particular with respect to fertilisation practices, as specified in the French "Agriculture Raisonnée" standards (Rosenberg and Gallot, 2002). The "Agriculture Biologique" (AB) scenario corresponds to organic agriculture according to the French version of European rules for organic animal production (Ministère de l'Agriculture et de la Pêche, 2000) and the European rules for organic crop production (CEE, 1991). The Label Rouge (LR) scenario corresponds to the "Porc Fermier Label Rouge" quality label, as specified in its rules of production (Groupements des fermiers d'Argoat, 2000). Impacts were referenced to two functional units referring to two farming system functions: units of pig produced (in kg live weight at slaughter) and units of land used (in ha).

Farming system scenarios

Data on crop production, transport distances, feed composition and system performance were based on statistics, estimates from experts and data from growers' associations. The feed compositions respected the specific rules of each production mode. According to the function and development stage of the animal, six diets were used for GAP (van der Werf et al., in press) and LR and five diets for AB.

For all crops, production corresponded to good agricultural practice, i.e. fertilisation according to anticipated crop needs and integrated pest management for GAP and LR. For the three scenarios, we assumed that pig manure (liquid manure for GAP, solid manure for LR, composted solid manure for AB) was used to fertilise Brittany-grown crops used as feed ingredients. The overall amount of manure or compost used for crop-based feed ingredients was adjusted, so as to correspond to the amount of manure that feeding the feed these ingredients were part of would yield. Additional N, P and K fertiliser was applied according to crop needs. For LR and GAP, yield levels were averages for 1996 – 2000 (AGRESTE, 2001; FAO, 2002). The yield levels of AB crops were defined according to the judgement of experts from the region the crops were produced in. Yields are lower than conventional yields: from -15% for maize to -40% for wheat and barley. For the processes concerning the transformation of crop products into feed ingredients and the production of feed, the inventory of resources used and emissions to the environment was limited to resources and emissions associated with the use of non-renewable energy. For ingredients resulting from processes yielding more than one product (e.g. soy cake, wheat gluten), resource use and emissions were allocated according to the economic value. Data for feed production (involving, amongst others: grinding, heating, mixing, pelleting) were from Sanders (2000).

We distinguished two stages in pig production: piglet production (PP) and weaning to slaughtering (WS). For GAP, data on technical performance for both PP and WS (Table 1) were according to published statistics (ITP, 2001). For LR, data concerning PP were from ITP (2001), data concerning WS were averages supplied by the LR producers' association. For AB, data on technical performance were based on an optimised model of organic pig production (Berger, 2000) adjusted according to expert judgement. For GAP and LR, manure was stored, while for AB, manure was composted, involving one or two turnings and taking 4 to 5 months.

Table 1. Characteristics of GAP, LR and AB pig farming systems.

	GAP*	LR*	AB*
Piglet production			
Housing	Slatted floor	Outdoor	Outdoor
Weaned piglet/productive sow/year	25.5	22.6	20.3
Weaning age, days	25.7	28	42
Surface per sow, m^2	<4	1000	1000
Feed per sow (boar included), kg/year	1313	1490	1695
Weaning to slaughtering			
Housing	Slatted floor	Straw litter	Straw litter
Surface per pig, m^2	0.85	2.6	2.3
Feed : gain ratio	2.7	2.9	3.2
Slaughter age, days	175	190	195
Slaughter weight, kg	113	115	120

*: GAP = Conventional production with good agricultural practice; LR = respects the "Porc Fermier Label Rouge" quality label; AB = organic agriculture

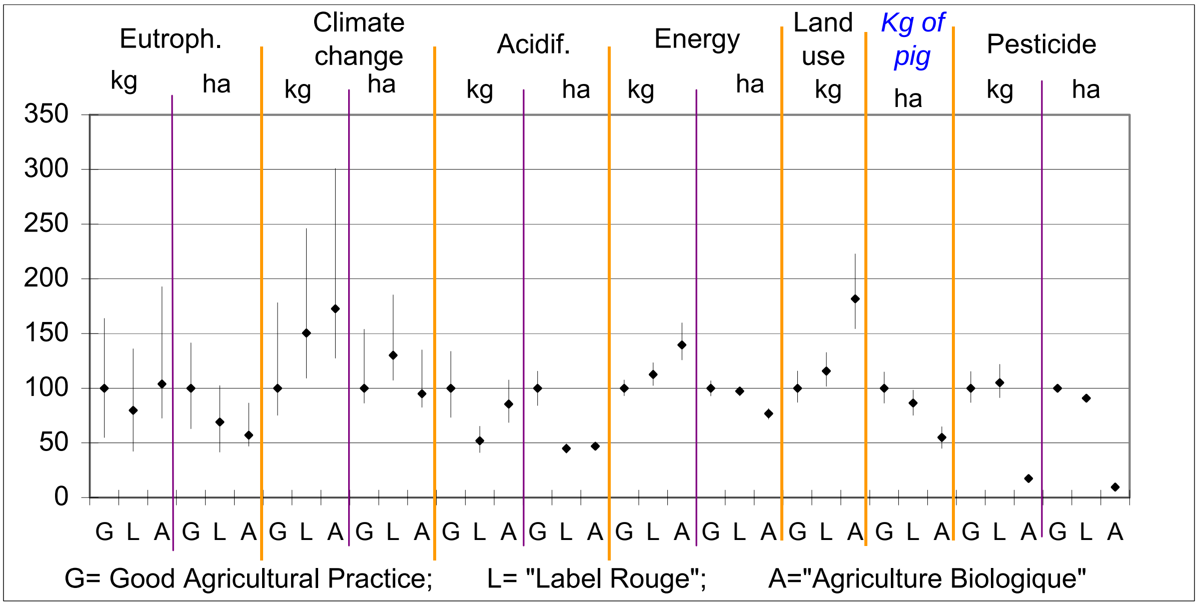


Figure 1. LCA results and estimated uncertainty for the GAP, LR and AB scenarios, expressed per kg of pig and per hectare as a percentage of the average result for GAP.

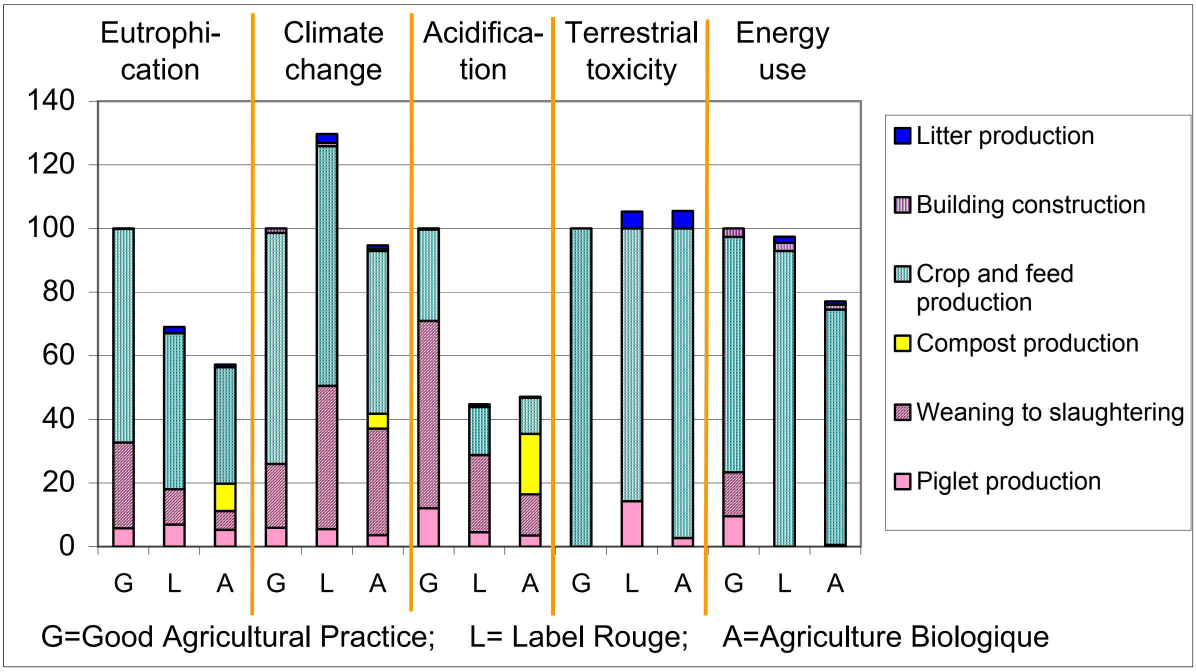


Figure 2. Contribution of six life cycle stages to five impact categories for the GAP, LR and AB scenarios, expressed per hectare and as a percentage of GAP.

Inventory data

Inventory data (resource use and emissions to the environment) were based on input-output data collected from different sources. Data associated with the production and delivery of inputs for crop production (fertilisers, pesticides, tractor fuel and machines) were derived according to Nemecek and Heil (2001). Data for energy carriers for road and sea transport were from the BUWAL 250 database (BUWAL, 1996). Data concerning resource use and emissions associated with buildings (production and delivery of materials, construction) were from Kanyarushoki (2001). Ammonia emissions due to the application of ammonium nitrate fertiliser were estimated according to ECETOC (1994) and ammonia emissions following application of slurry were according to Morvan et Leterme (2001). Ammonia and nitrous oxide emissions from slurry in pig buildings were from IPCC (1996) and UNECE (1999). Methane emissions due to enteric fermentation and housing type were from IPCC (1996). Data on the production of excreta, emissions from buildings, during storage, during composting and from crops and paddocks, were chiefly obtained with the support of an expert panel from the Institut National de la Recherche Agronomique. The panel comprised: J. Y. Dourmad, Th. Morvan, J.M. Paillat, P. Robin and F. Vertès. The panel based its expertise on their experiments, simulation models and on their interpretation of the available literature. The environmental impacts due to the slurry or the manure applied on the crops were associated with the crop production.

Uncertainty analysis
In order to explore the robustness of our results, an uncertainty analysis was conducted. Crop yields, WS feed to gain ratio, field emissions (NH_3, N_2O and NO_3) and emissions of NH_3 and N_2O from buildings, manure storage and composting were identified as important issues for the variability of results. For each emission and parameter concerning these issues, specific high and low values reflecting what we coined "realistic" rather than overall variability were defined in addition to the default reference value. The intervals thus defined were much larger for emissions at field level (on average –40% and +83%) than for emissions in buildings (on average ±18%) and for technical parameters (±6% for feed to gain ratio and around ±10-15% for crop yield). The "realistic" uncertainty interval thus defined contains about two thirds of the overall variability for the parameter concerned. We combined all favourable values (better performance, lower emissions) for key-parameters to obtain a favourable variant of each scenario and similarly we combined all unfavourable values (poorer performance, higher emissions) to obtain an unfavourable variant of each scenario. In addition to the default scenario based on the reference values, these two variants are proposed as indicators of overall uncertainty for each scenario.

Impact assessment
The environmental impact categories considered in this study are: eutrophication (in kg PO_4-eq), climate change (in kg CO_2-eq), acidification (in kg SO_2-eq), terrestrial toxicity due to heavy metal accumulation (in kg 1.4-dichlorobenzene-eq), energy use (in MJ Lower Heating Value (LHV)-eq), land use (in m^2.year) and pesticide use (in kg of active ingredient). Eutrophication, acidification, terrestrial ecotoxicity potentials were calculated using characterisation factors by Guinée et al. (2002). Global warming Potential for a 100 year time horizon (GWP100) was calculated according to the GWP100 factors by IPCC (Houghton et. al., 1996). Energy use was calculated using the LHV proposed in the SIMAPRO 1.1 method (PRé Consultants, 1997). Pesticide use (in kg active matter used) refers to the global quantity of pesticide used for crop production.

Results
Per kg of pig produced, eutrophication was smallest for LR (0.0166 kg PO_4-eq) followed by GAP (0.0208) and AB (0.0216). Per ha, eutrophication was largest for GAP (38.3 kg PO_4-eq) intermediate for LR (26.4) and smallest for AB (22.9). Both per kg and per ha, overall uncertainty was large and asymmetrical, particularly for AB (Fig. 1). From 64% (AB) to 71% (LR) of eutrophication was due to crop and feed production. 10% (AB) to 27% (GAP) was due to weaning to slaughtering and 6% (GAP) to 10% (AB) was due to piglet production because of emissions of NH_3 and N_2O in buildings. For AB, 15% was due to compost production. Litter production accounted for 1.3% (AB) and 2.8% (LR), building construction contributed 0.1% (GAP, LR, AB) (Fig. 2).

Per kg of pig, climate change was 2.3 kg CO_2-eq for GAP, 3.46 for LR and 3.97 for AB. Per ha, climate change was larger for LR (5510 kg CO_2-eq) than for AB (4022) and GAP (4236).

Both per kg and per ha uncertainty intervals were very large and asymmetrical for the three scenarios (Fig. 1). From 54% (AB) to 73% (GAP) of climate change was due to crop and feed production, 20% (GAP) to 35% (LR, AB) was due to weaning to slaughtering, 4% (LR, AB) to 6% (GAP) to piglet production and for AB 5% was due to compost production. Litter production accounted for 1.2% (AB) and 2.1% (LR) and building construction for 0.7% (AB) to 1.4% (GAP) (Fig. 2).

Acidification per kg of pig was 0.0226 kg SO_2-eq for LR, 0.0372 for AB and 0.0435 for GAP. Acidification per ha was very close for LR (36.0) and AB (37.7) and much larger (80.1) for GAP. Both per kg and per ha, uncertainty intervals for acidification were smaller than for eutrophication and climate change, in particular when expressed per ha for LR and AB (Fig. 1). Pig production (PP and WS) was the main contributor to acidification for GAP (71%) and LR (64%), while for AB, compost production contributed more (40%) than pig production (35%). Crop and feed production accounted for 24% (AB) to 34% (LR), litter production accounted for 0.5% (AB) and 1.4% (LR), building construction accounted for 0.3% (GAP, AB) to 0.6% (LR) (Fig. 2).

Per kg of pig, terrestrial toxicity was 0.0165 kg 1.4-dichlorobenzene-eq for GAP, 0.0184 for LR and 0.0304 for AB. Per ha the three scenarios had similar impacts. Per kg, the uncertainty intervals were relatively small, expressed per ha, the intervals were very small or non-existent (not shown). Crop and feed production was the main source of terrestrial toxicity for LR (81%) and AB (92%) and the only one for GAP. For LR, (outdoor) piglet production contributed 14% and 2.6% for AB. Litter production contributed 5% for LR and AB (Fig. 2).

Energy use per kg of pig was close for GAP (15.9 MJ) and for LR (17.9), for AB, energy use was larger (22.2). Conversely, energy use per ha was close for GAP (29282 MJ) and for LR (28503) but smaller for AB (22492). The uncertainty intervals expressed per kg were small, expressed per ha they were very small (Fig. 1).

Crop and feed production was the main contributor to energy use, ranging from 74% for GAP to 96% for AB. For GAP, PP plus WS contributed for 23% while no energy was used for these stages for LR and AB. Building construction accounted for 2.0% (AB) to 2.7% (GAP) of energy use. Straw litter production accounted for 1.3% (AB) to 1.9% (LR) (Fig. 2).

Land use per kg of pig was larger for AB (9.87 m^2.year) than for GAP (5.43) and LR (6.28). The uncertainty intervals were relatively small (Fig. 1). Crop and feed production was the most important contributor to land use, ranging from 89% (LR) to 100% (GAP) (not shown). Outdoor piglet production accounted for 6% (AB) and 8% (LR) of land use. Litter production accounted for 3.1% (AB) and 3.4% (LR) of land use (not shown). With respect to the amount of pig per ha, GAP was the most productive scenario (1840 kg/ha) followed by LR (1590) and AB (1010) (Fig. 1).

Pesticide use per kg of pig was close for GAP (1.37 g) and LR (1.44) and much smaller for AB (0.239 g). Per ha it was close for GAP (2.52 kg/ha) and LR (2.29) and much smaller for AB (0.24 kg/ha). The uncertainty intervals expressed per kg were relatively small, expressed per ha, the intervals were very small or non-existent. For GAP and AB, crop and feed production was the sole contributor to pesticide use. For LR, litter production contributed 4% (not shown).

Comparison of the three scenarios for impacts and yield

Current intensive pig production systems have a poor image with the general public because they are associated with environmental pollution (in particular poor water quality) on a regional scale (Petit and van der Werf, 2003). Given this perspective, we based our examination of the three scenarios primarily on their impacts and productivity per ha. Expressed per ha, LR had lower impacts than GAP for eutrophication and acidification, a higher impact for climate change, the other impacts were similar for GAP and LR; LR was less productive (- 14%) (Fig. 1). Expressed per ha, AB had lower impacts than GAP for eutrophication, acidification, energy use and pesticide use, for climate change and terrestrial toxicity impacts were similar; pig production was 45% less for AB than for GAP (Fig. 1).

The lower impacts per ha of LR and AB relative to GAP for eutrophication and acidification resulted from lower emissions of eutrophying and acidifying substances from the weaning to slaughtering stage in straw-litter buildings (despite compost-related emissions for AB) and from the crop and feed production stage (Fig. 2). Pesticide use per ha was much lower for AB but not zero as the feed contained 10% of non-organic ingredients.

Hot-spots and margin for improvement

The GAP scenario did worse than LR and AB for eutrophication and acidification. NO_3 (crop and feed production) and NH_3 (crop and feed production and pig production: PP and WS) were the predominant substances contributing to these two regional impacts. As fertilisation was optimised in the GAP scenario, the systematic introduction of catch crops (rather than 50%) seems the most promising measure to reduce nitrate emissions. Although the field application of slurry was relatively optimised for GAP, ammonia losses can be further reduced by adopting slurry injection technology. Decreasing dietary protein can lead to significant reduction of ammonia emission along the building, storage and application stages (Portejoie et al., 2002). Similarly, the coverage of slurry stores can significantly reduce ammonia emissions during storage and application (Portejoie et al., 2003). Pesticide use was another major drawback of the GAP scenario.

The LR scenario contributed more to climate change than GAP and AB, due to the WS stage, when pigs are in a straw litter building. We identified N_2O, a very potent greenhouse gas, as the predominant substance contributing to climate change for this stage. Litter-based housing systems suffer from a low level of standardisation. So far, the type of litter (Robin et al., 1999), litter management (Kermarrec, 1999) and animal density (Robin, 2002) have been identified as influent parameters affecting gaseous emissions in litter systems. As for GAP, a low-protein diet could help reduce emissions for LR. The LR scenario thus seems to have a large margin of improvement. As for GAP, pesticide use is a drawback of LR.

Expressed per ha, AB had lower acidification and eutrophication impacts than GAP, expressed per kg, AB and GAP were similar for these impacts. For AB, 15% of eutrophication and 40% of acidification were due to compost production, which thus constitutes a major disadvantage for AB. However, the emissions associated with compost production can be reduced by adapting the heap composition (Ekinci et al., 2000; Sommer and Møller, 2000) and by optimising composting techniques such as reducing the turning frequency or covering the heap (Robin et al., 2001). As for LR, the AB straw litter system presents a major contribution to climate change. Like GAP and LR, AB seems to have an important margin of improvement. Finally, in certain contexts, the low productivity per ha of AB might be a limitation to its development.

Conclusion

Our comparison of three contrasting and relatively optimised scenarios of pig production produces useful knowledge for decision makers at different levels. For each scenario, hot-spots as well as important margins of improvement have been identified. The use of favourable and unfavourable scenarios allowed an estimation of the uncertainty of the results.

References

AGRESTE, 2001. Tableaux de l'Agriculture Bretonne 2001. Direction Régionale de l'Agriculture et de la Forêt, Service Régional de Statistique Agricole, Rennes, France.

Berger, F., 2000. Les défis du porc biologique. Chambre régionale d'agriculture des Pays de Loire. France.

BUWAL, 1996. Ökoinventare für Verpackungen. Schriftenreihe Umwelt Nr. 250/1+2, Bundesamt für Umwelt, Wald und Landschaft, Bern, Switzerland.

CEE, 1991. Règlement (CEE) n°2092/91 du conseil du 24 juin 1991 concernant le mode de production biologique de produits agricoles et sa présentation sur les produits agricoles et les denrées alimentaires (*JO* L 198 du 22.7.1991).

ECETOC, 1994. Ammonia emissions to air in western Europe. Technical report no. 62. European Chemical Industry Ecology & Toxicology Centre, Brussels, Belgium.

Ekinci, K., Keener, H.M., Elwell, D.L., 2000. Composting short paper fiber with broiler litter and additives. Part 1: effects of initial pH and carbon/nitrogen ratio on ammonia emission. *Compost Sciences and Utilization* **82**: 160 – 172.

FAO, 2002. FAOSTAT Agriculture Data. Available at : http://apps.fao.org/.

Groupement des fermiers d'Argoat, 2000. Cahier des charges éleveur – porcs fermiers élevés en plein air. Homologation n°20-88. Saint-Brieuc, France.

Guinée, J.B., Gorrée, M., Heijungs, R., Huppes, G., Kleijn, R., de Koning, A., van Oers, L., Wegener Sleeswijk, A., Suh, S., Udo de Haes, H.A., de Bruijn, H., van Duin, R., Huijbregts, M.A.J., 2002. Life cycle assessment. An operational guide to the ISO standards. Centre of Environmental Science, Leiden University, Leiden, The Netherlands.

Houghton, J.T., Meira Filho, L.G., Callander, B.A., Harris, N., Kattenberg, A., Maskell, K., 1996. Climate change 1995: the science of climate change. Cambridge University Press, Cambridge, United Kingdom.

Huijbregts M.A.J., Norris, G., Bretz, R., Ciroth, A., Maurice, B., von Bahr, B., Weidema, B., de Beaufort A.S.H., 2001. Framework for Modelling Data Uncertainty in Life Cycle Inventories. *Int. J. LCA* **6** (3): 127 – 132.

IPCC, 1996. Intergovernmental Panel on Climate Change. Revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories : Reference Manual (3). Available at : http://www.ipcc-nggip.iges.or.jp/public/gl/invs6.htm.

ITP, 2001. Résultats Gestion Technico-économique. ITP, Paris, France.

Kanyarushoki, C., 2001. Evaluation de l'impact environnemental des bâtiments porcins – Application de la méthode d'analyse du cycle de vie. Mémoire de fin d'études DESS. Université Rennes 1, Rennes, France.

Kermarrec, C., 1999. Bilan et transformations de l'azote en élevage intensif de porcs sur litière. PhD thesis. Université de Rennes I, Rennes, France.

Ministère de l'Agriculture et de la Pêche, 2000. Cahier des charges concernant le mode de prodution et de préparation biologique des animaux et des produits animaux définissant les modalités d'application du règlement CEE n° 2092/91 modifié du conseil et/ou complétant les dispositions du règlement CEE n° 2092/91. Homologué par l'arrêté interministériel du 28 août 2000. *J.O.R.F.* du 30 août 2000. Direction des politiques économique et internationale, Paris, France.

Morvan, Th., Leterme, Ph., 2001.Vers une prévision opérationnelle des flux d'azote résultant de l'épandage de lisier : paramétrage d'un modèle dynamique de Simulation des transformations de l'azote des lisiers (STAL). *Ingénieries* **26**: 17 – 26.

Nemecek, T., Heil, A., 2001. SALCA – Swiss Agricultural Life Cycle Assessment Database. Version 012, December 2001. FAL, Swiss Federal Research Station for Agroecology and Agriculture, Zuerich, Switserland.

Petit, J., van der Werf, H M G, 2003. Perception of the environmental impacts of current and alternative modes of pig production by stakeholder groups. *Journal of Environmental Management* 68: 377-386.

Portejoie, S., Dourmad, J-Y., Martinez, J., Lebreton, Y., 2002. Effet de la réduction du taux protéique de l'aliment sur la volatilisation ammoniacale des effluents porcins. *Journées Rech. Porcine en France* **34**: 167-174.

Portejoie, S., Martinez, J., Guiziou, F., Coste, C.M., 2003. Effect of covering pig slurry stores on the ammonia emission processes. *Bioresource Technology* **87**: 199-207.

PRé Consultants, 1997. SimaPro 2 method. In: Database Manual. Pré Consultants B.V., Amersfoort, The Netherlands.

Robin, P., 2002. Maîtrise des émissions gazeuses en bâtiments sur litière (validation en élevages des résultats acquis en station expérimentale). Rapport final première année (2002). Convention DERF-INRA n°A 01592-2001-2002. Rennes, France.

Robin, P., De Oliveira, P.A., Kermarrec, C., 1999. Productions d'ammoniac, de protoxyde d'azote et d'eau par différentes litières de porcs durant la phase de croissance. *Journées Rech. Porcine en France* **31**: 111-115.

Robin, P., Aubert, C., Bline, D., Tricot, G., Vacher, V., 2001. Maîtrise du compostage de fumier de volailles à la ferme : mélange initial, retournements, couvertures. *Sciences et Techniques Avicoles* **35**: 9-12.

Rosenberg, P.E., Gallot, J., 2002. Référentiel de l'agriculture raisonnée. Arrêté du 30 avril 2002 relatif au référentiel de l'agriculture raisonnée. *Journal Officiel de la République Française* **104** (4 mai 2002): 8519.

Sanders, J., 2000. L'evaluation des systèmes de production animale: l'Analyse de Cycle de Vie peut-elle servir à construire des indicateurs environnementaux pour l'alimentation porcine ? INRA, UMR SAS, Rennes, France.

Sommer, S.G., Møller, H.B., 2000. Emission of greenhouse gases during composting of deep litter from pig production – effect of straw content. *Journal of Agricultural Science, Cambridge* **134**: 327-335.

UNECE, 1999. Control options/techniques for preventing and abating emissions of reduced nitrogen compounds. United Nations Economic and Social Council, Economic Commission for Europe.

Van der Werf, H.M.G., Petit, J., 2002. Evaluation of the environmental impact of agriculture at the farm level: a comparison of twelve indicator-based methods. *Agriculture, Ecosystems and Environment* **93** (1): 131 – 145.

Van der Werf, H.M.G., Petit, J., Sanders, J. The environmental impacts of the production of concentrated feed : the case of pig feed in Bretagne. *Agricultural Systems*, in press.

LCA of Animal Products from Different Housing Systems in Switzerland: Relevance of Feedstuffs, Infrastructure and Energy Use

Erzinger, S., Dux, D., Zimmermann, A. and Badertscher Fawaz, R.*

**Agroscope FAT Taenikon, Swiss Federal Research Station for Agricultural Economics and Engineering, CH-8356 Ettenhausen, Switzerland. Phone +41 52 3683239, fax +41 52 3651190, stefan.erzinger@fat.admin.ch*

Abstract

For many foodstuffs, especially animal products, agricultural production is an environmentally relevant production step. Milk and meat are produced in many different housing systems. The differences in the environmental impacts of these production systems are as yet little known. The goal of our study was to assess the environmental impacts of conventional and animal-friendly housing systems in Switzerland. We compared case studies of two housing systems for dairy cows and for fattening pigs. The most important factor proved to be the feeding regime. The building infrastructure was also relevant, especially for energy consumption and human toxicity. Improvements in the choice and production of feedstuffs should be introduced. Possible measures are low-emission fertilisation, efficient mechanisation, short transport distances and little drying.

Keywords: animal farming, housing system, life cycle assessment, milk, pork

Introduction

Animal-friendly housing is finding increasing practical application in Switzerland, partly as a result of animal welfare legislation, partly within the framework of direct payments for especially animal-friendly husbandry or in the context of various label schemes. Not much information is available on the environmental impacts of different animal housing systems, data gaps being particularly common in relation to new types of housing. The goal of the study was to compare the environmental impacts of various animal housing systems for dairy cows and fattening pigs using life cycle assessments.

Methods and systems investigated

Each of the following systems were compared for different herd sizes and feed variants:

- Milk from cows in "cubicle housing" and "tied housing" and
- Fattening pigs from "multi-surface systems with littered lying area and exercise yard" and from "pens with fully slatted floors".

Special emphasis was placed on an accurate study of the buildings. This includes construction (building materials, construction processes), operation, maintenance and dismantling of the buildings. Service life of the buildings is 50 years. Building parts and equipment with a shorter life time are replaced during this 50 years (Nemecek et al., 2003)

The comparison of these different animal housing systems was carried out using production processes which were determined on the basis of empirical values and expert know-how. Additionally, the pork case study is based on a survey of around 90 fattening pig farms. All the influencing variables unconnected with housing type (e.g. mechanisation) were identically defined where possible. However there were also differences between the processes resulting not from housing type but from labour organisation and farm management. The processes were not defined and optimised with reference to ecological criteria, but illustrated conditions normally found on the farm. The study should therefore be regarded as a case study.

The farm gate was the system boundary for assessment. All preliminary stages and infrastructure (buildings, machinery) were included. The processing and distribution of the milk and fattened pigs did not form part of the study.

Direct emissions of ammonia, methane and nitrous oxide from animal metabolism, housing areas and slurry storage were difficult to assess. No reliable data on emissions were available, particularly for the animal-friendly systems. Four scenarios were therefore defined for dairy cattle and for fattening pigs (Tab. 1). The scenarios "Standard1" and Standard2" base on current emission factors published in Switzerland and Germany respectively. The scenario "High" uses very high emission factors, whereas the scenario "Low" uses very low factors. These two scenarios do not distinguish between housing systems. The intention of the scenarios "High" and "Low" is to get a range for the relevance of direct emissions from the housings compared to the total emissions from the entire systems.

Table 1. Scenarios for direct emissions from animal housing (incl. slurry pit) for dairy cows and fattening pigs [kg gas/(place×year)]. TH = tied housing, CH = cubicle housing; FSF = fully slatted floors, MSS = multi-surface system.

	Dairy cows					
	NH_3		**CH_4**		**N_2O**	
Scenario	TH	CH	TH	CH	TH	CH
Standard1 [1]	11.2	22.9	186	183	0.5	1.0
Standard2 [2]	6.3	16.7	126	126	0.7	1.3
High [3]	41.5	42.1	240	240	1.4	1.8
Low [3]	3.5	4.1	50	50	0.4	0.8
	Fattening pigs					
	NH_3		**CH_4**		**N_2O**	
Scenario	FSF	MSS	FSF	MSS	FSF	MSS
Standard1 [1]	2.3	2.3	4.7	5.1	0.05	0.05
Standard2 [2]	3.8	5.0	4.0	2.5	0.12	0.12
High [3]	7.1	7.1	7.0	7.0	0.52	0.52
Low [3]	1.1	1.1	0.5	0.5	0.02	0.02

1: Ammonia (NH_3): Menzi et al. (1997); Methane (CH_4): Minonzio et al. (1998); Nitrous oxide (N_2O): Schmid et al. (2000); all in Nemecek (2002)
2: UBA (2002), adapted
3: own assumptions based on analysis of the literature

Twelve environmental impacts were calculated. The impacts were assessed according to Rossier and Gaillard (2001) and Nemecek et al. (2003a). Neither noise, odour nor impact on biodiversity, soil fertility or landscape were studied. Life cycle assessment calculations were carried out using TEAM software (Version 3.0) and the SALCA life cycle inventory database of the Swiss Agricultural Research Stations (Nemecek, 2002). Only three of the twelve environmental impacts investigated will be discussed here, namely energy consumption, eutrophication and ecotoxicity. These indicators have proved particularly meaningful in previous studies (Rossier and Gaillard, 2001).

Milk case study

Milk is the most important raw product of Swiss farming and is frequently produced in intensive grassland-based systems. Production is organised on a small scale, many farms owning around 20 cows. Although tied housing systems are widespread (approx. 80%), new buildings are almost exclusively loose housing. The systems studied are described in Table 2.

Table 2. Description of the types of dairy cattle housing studied.

		Tied housing	Cubicle housing
Herd size		20 and 40 dairy cattle places	
Functional unit		1 litre cooled milk from farm tank	
By-products [kg live weight/(cow×y)]			
• Starter calves		40.5	44.3
• rearing calves		58.8	54.3
• culled cattle		205.0	190.3
Milk yield [1] [kg ECM/(cow×y)]		6987	7103
Feed	without silage [2]	*hay, grass, fodder beet, maize cubes, concentrate, pasture*	
	with silage	grass silage, maize silage, grass, hay, concentrate, pasture	
Pasture [3]		60 half days pasture	198 half days pasture
Building		with exercise yard	
• Ventilation		gravity flues	free ventilation
• Milking		pipeline milking system	herringbone milking parlour

1: Difference due to higher fertility and longer useful life in loose housing (Badertscher, 2003)
2: Prerequisite for the production of cheese made from raw milk
3: Tied housing: minimum animal welfare legislation; cubicle housing: full growing season

Environmental impact was split between milk and by-products (Tab. 2) using a financial allocation. Under this method, the milk accounted for 88% of the environmental impact.

Results

Table 3 and Figure 1 show the results of the housing systems, herd sizes and feed variants studied.

Table 3. Environmental impacts of milk from different dairy cow housing systems and feed variants (per kg cooled milk at farm gate). TH = tied housing, CH = cubicle housing.

	Herd size	**20 cow places**				**40 cow places**			
	Feed	without silage		with silage		without silage		with silage	
	Housing type	TH	CH	TH	CH	TH	CH	TH	CH
Energy	[MJ_{eq}/kg]	6.0	5.8	4.6	4.4	5.7	5.4	4.4	4.1
Eutrophication	[gPO_{4eq}/kg]	4.2	4.5	3.7	4.1	4.2	4.5	3.7	4.1
Ecotoxicity	[gZn_{eq}/kg]	0.21	0.21	0.21	0.20	0.20	0.20	0.21	0.20

The environmental impact of milk production is determined mainly by the feedstuffs (Fig. 1), which contribute between 50 and 85% to the three impacts analysed. This includes the agricultural production, processing (e.g. hay aeration) and transports. The buildings (construction, operation and dismantling) play a significant role in energy consumption with a share of approximately 25%.

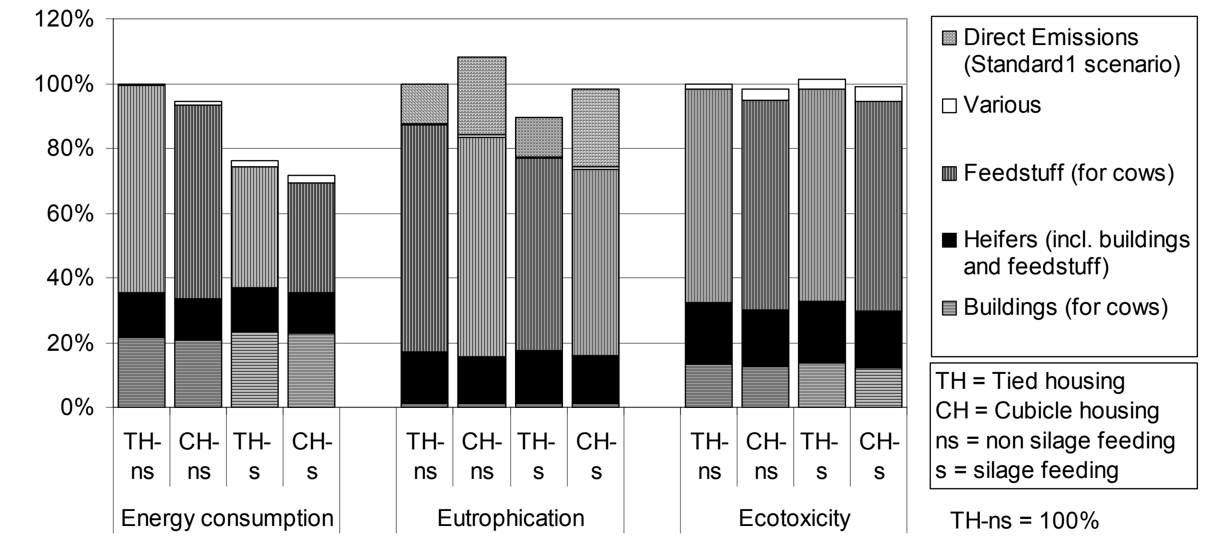


Figure 1. Comparison of the environmental impact of milk from tied housing and cubicle housing with 40 cow places each.

The variants with silage feed needed 24% less energy than those without silage. This difference is above all due to hay ventilation and drying of maize cubes in the non-silage feeding variant. Average energy consumption was 5 MJ_{eq} per kilo of milk. Compared with the metabolisable energy of 2.7 MJ/kg milk, primary energy input was thus almost twice as high as energy output for human nutrition.

Maize cubes (chopped, dried and pressed maize for non-silage feeding) and concentrate (cereals) were the feedstuffs with a high contribution to the three environmental impacts analysed. Hay aeration used around 10% of total energy consumption in the variants without silage feed. The impact of silage, grass, pasture and fodder beet was low. The environmental impact of feedstuffs was determined mainly by agricultural production and drying. Transport and processing were of less importance.

Herd size barely affected environmental impact of the entire system as well as of the buildings used. Both herd sizes studied showed a similar impact per cow place. The housing type only affected direct emissions, other environmental impacts were similar. Direct emissions from housing and slurry pit were caused mainly by ammonia emissions, accounting for between 3 and 40% of total eutrophication, depending on the emission scenario (Tab. 4). In the standard scenarios, the direct emissions from housing were around twice as high for cubicle housing as for tied housing.

Table 4. Influence of scenarios for direct emissions on „eutrophication“ environmental impact for the „40 places without silage feed“ variant. TH = tied housing, CH = cubicle housing. For definition of scenarios see Tab. 1.

	Scenario	**Standard1**		**Standard2**		**High**		**Low**	
	Housing type	TH	CH	TH	CH	TH	CH	TH	CH
Eutrophication	[gPO_{4eq}/kg milk]	4.2	4.7	4.0	4.4	6.1	6.0	3.8	3.8
- of which direct emission from housing		0.5	1.0	0.3	0.8	2.4	2.3	0.1	0.1
Share direct emission from housing	[%]	12	22	8	17	40	39	3	3
Share other system	[%]	88	78	92	83	60	61	97	97

Discussion

In particular the type of feedstuff (including cultivation, conservation, processing and transports) determines the potential environmental impact of milk production. The way to reduce environmental impact is to optimise roughage production (Nemecek and Huguenin, 2002). Possible measures are:

- extensive roughage production with low fertiliser use
- more pasture, less grass harvesting
- silage or field dried hay instead of hay aeration and maize drying. Use of renewable energy carriers in hay aeration.

There was hardly any difference in the environmental impact of the building infrastructure and usage in the two dairy housing systems studied. The environmental soundness of dairy housing may be optimised by ensuring minimum possible energy consumption during milk production and cooling. Direct emissions from the cowsheds could be expected to differ between the systems, but these cannot be quantified using the data available. Research is needed here.

Other studies often take no account of the infrastructure, and the basic data and calculation methods are seldom accurately described. Comparisons are therefore difficult. Compared with the mean values of 35 dairy farms from Rossier and Gaillard (2001), the energy and eutrophication values calculated here are roughly one third lower, but twice as high for ecotoxicity – although they fall within the variability noted there. The figures for primary energy consumption are higher than in Cederberg (1998), Haas et al. (2001) or Hogaas Eide (2002).

Pork case study

Two herd sizes and feed variants were studied for pork (Tab. 5). Pig fattening gives rise to no by-products, so no allocation was required. The total environmental impacts were down to the animals produced.

Table 5. Description of the fattening pig housing systems studied.

		Fully slatted floors	**Multi-surface system**
Herd size		300 and 1000 fattening pig places	
Functional unit		1 kg pig (live weight at farm gate)	
• Daily gain [kg/day]		0.748	0.757
• Days to reach slaughter weight		111	107
• feed conversion [MJ DE per kg live weight gain]		36	39
Feed	Complete diet	Complete diet	
	Whey	Whey, energy and protein supplementary feed, fat	–
Building		*No exercise yard*	With exercise yard and littered lying area.
• Ventilation		*Forced ventilation*	Free ventilation
• Heating		Electric radiators	No heating

Results

The results of the housing systems, herd sizes and feed variants studied are shown in Table 6 and Figure 2.

Table 6. Environmental impact of different housing systems and feeding variants for fattening pigs (per kg live weight at farm gate). FSF = fully slatted floors, MSS = multi-surface system.

	Herd size	**300 pig places**			**1000 pig places**		
	Feed	Whey	Complete diet		Whey	Complete diet	
	Housing system	FSF	FSF	MSS	FSF	FSF	MSS
Energy	[MJ_{eq}/kg	30.7	33.3	27.8	27.7	30.2	27.6
Eutrophication	[gPO_{4eq}/kg	28.8	36.2	37.5	29.9	37.3	38.5
Ecotoxicity	[gZn_{eq}/kg	0.80	0.75	0.78	0.81	0.76	0.81

Feedstuff provision is by far the greatest factor affecting the environmental impact of pig fattening (Fig. 2). Depending on the type of environmental impact, the feedstuffs used in fattening and piglet production account for up to 90% of the environmental impact.

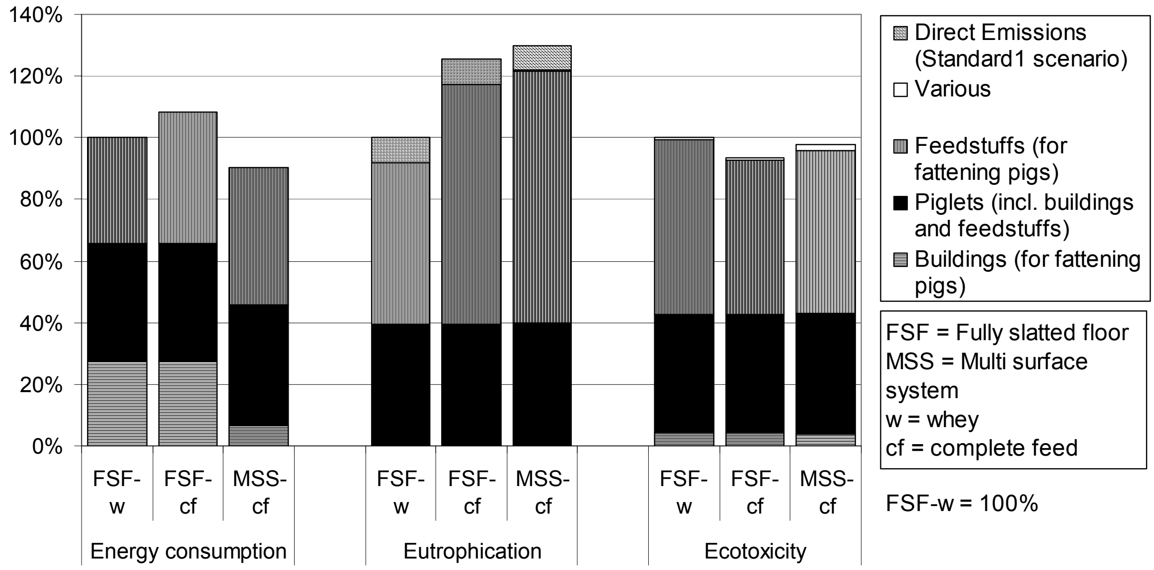


Figure 2. Comparison of the environmental impact of pork from fully slatted and multi-surface housing with two feed variants and herd size of 300 fattening pigs.

The buildings have a relevant influence on energy consumption, in housing with fully slatted floors the buildings account for almost 30% of energy consumption.

Direct emissions from the fattening pigs housing contributed roughly 5% to eutrophication in the scenario “Low” and around 35% in the scenario “High”.

Herd size had a limited effect on energy consumption. Although the building-related energy consumption per kilogram of meat produced did fall as herd size increased, the energy consumption for the supply of feedstuffs was unchanged. Herd size also barely affected any of the other environmental impacts studied.

Discussion

The feedstuffs used have a crucial effect on the potential environmental impact of pork production. Good feed utilisation and the use of feedstuffs produced by environmentally sound methods are therefore of primary importance in reducing environmental pollution. The key factors here are:

- Agricultural production using low-emission fertiliser as well as efficient mechanisation,
- Little transportation and drying and

- Use of by-products from milk processing, milling, sugar and oil production or other industrial processes – provided that these products are not contaminated with pollutants or competing with other fields of application.

Conclusions

The results shown here provide pointers to the environmental impact of different housing systems for fattening pigs and dairy cattle.

The differences found were caused not only by the actual housing system, but in particular by differences in the feeding regime. The present results should therefore always be interpreted as the combined result of feeding regime and housing system.

Feedstuff provision was central – both to environmental impact and production cost (Badertscher, 2003).

Infrastructure (buildings and installations) had a significant effect on the overall environmental impact of animal production. It should therefore definitely be taken into account by LCAs. Herd size, on the other hand, had little effect on the results. There are large data gaps in the assessment of direct emissions from housing, an influence on eutrophication.

LCAs are well suited to analysing the environmental impact of entire animal production systems, assuming the availability of high quality LCI databases. As shown above, the most important factor in optimising animal production is to provide environmentally sound feedstuffs. Here we should also be dealing with biodiversity and landscape issues beyond the LCA currently used.

Acknowledgment

The project was funded by the Swiss Federal Veterinary Office. Issues of economic viability, workload, animal health and product quality were studied in addition to the environmental life cycle assessment discussed here (Badertscher, 2003).

Literature

Badertscher, R., 2003. Tierwohl und Wirtschaftlichkeit: Ein Widerspruch? Paper to be presented at the Gewisola Tagung Hohenheim 2003 Hohenheim (Germany), Sept 29 - Oct 01, 2003.

Cederberg, C., 1998. Life Cycle Assessment of Milk Production. A Comparison of Conventional and Organic Farming. SIK-Document 143. Swedish Institute for Food and Biotechnology, Gothenburg.

Erzinger, S. and Badertscher Fawaz, R., 2001. Life Cycle Assessment of animal housing systems as part of an overall assessment. Proceedings International Conference on LCA in Foods, p. 118-122. SIK-Report Nr. 643. Swedish Institute for Food and Biotechnology, Gothenburg.

Haas, G., Wetterich, F. and Köpke, U., 2001. Comparing intensive, extensified and organic grassland farming in southern Germany by process life cycle assessment. *Agriculture, ecosystems and environment* **83**, 43-53.

Hogaas Eide, M., 2002. Life cycle assessment (LCA) of industrial milk production. *Int J LCA* **7** (2), 115-126.

Menzi, H., Frick, R. and Kaufmann, R., 1997. Ammoniak-Emissionen in der Schweiz: Ausmass und technische Beurteilung des Reduktionspotentials. Schriftenreihe der FAL 26. Swiss Federal Research Station for Agroecology and Agriculture, Zurich-Reckenholz.

Minonzio, G., Grub, A. and Fuhrer, J., 1998. Methan-Emissionen der schweizerischen Landwirtschaft. Schriftenreihe Umwelt Nr. 298. Swiss Agency for the Environment, Forests and Landscape (SAEFL), Berne.

Nemecek, T., Heil, A., Huguenin, O., Meier, S., Erzinger, S., Blaser, S., Dux, D. and Zimmermann, A., 2003. Life Cycle Inventories of Agricultural Production Systems. Final report ecoinvent 2000 No. 15. FAL Reckenholz, FAT Tänikon, Swiss Centre for Life Cycle Inventories, Dübendorf, CH, retrieved from: www.ecoinvent.ch.

Nemecek, T., Heil, A., Erzinger, S. and Zimmermann, A., 2003a. SALCA Swiss Agricultural Life Cycle Assessment Database. Version 031. Swiss Federal Research Station for Agroecology and Agriculture, Zurich-Reckenholz.

Nemecek, T., 2002. SALCA-Templates. Swiss Agricultural Life Cycle Assessment Database. Beschreibung der Mustersysteme „Betrieb“ und „Kultur“, Version 1.20. Swiss Federal Research Station for Agroecology and Agriculture, Zurich-Reckenholz.

Nemecek, T. and Huguenin, O., 2002. Beurteilung von Graslandsystemen. Schriftenreihe der FAL 38, 16-18. Swiss Federal Research Station for Agroecology and Agriculture, Zurich-Reckenholz.

Rossier, D. and Gaillard, G., 2001. Bilan écologique de l'exploitation agricole. Méthode et application à 50 entreprises. Swiss Federal Research Station for Agroecology and Agriculture, Zurich-Reckenholz (available from bibliothek@fal.admin.ch).

Schmid, M., Neftel, A. and Fuhrer, J., 2000. Lachgasemissionen aus der Schweizer Landwirtschaft. Schriftenreihe der FAL 33. Swiss Federal Research Station for Agroecology and Agriculture, Zurich-Reckenholz.

UBA (eds.), 2002. BMVEL/UBA-Ammoniak-Emissionsinventar der deutschen Landwirtschaft und Minderungsszenarien bis zum Jahre 2010. German Federal Environmental Agency, Berlin.

Qualitative Life Cycle Assessment of Thai Shrimp Product

Mungkung, R. and Clift, R.
Corresponding address: Centre for Environmental Strategy (CES), School of Engineering, University of Surrey, Guildford, Surrey GU2 7XH, UK Phone: +44 (0)1483 689559 Fax: +44 (0)1483 686671 E-mail: r.mungkung@surrey.ac.uk

Abstract

Economic and social importance of shrimp farming is evident in terms of export values, income generation, and employment opportunity. However, the continual rise of shrimp consumption and proliferation of shrimp farms has become controversial because of their associated potential environmental impacts. The impacts of shrimp farming have been highlighted and their sustainability has been under criticism. Much attention has been paid to Thailand as the world's leading producer of farmed shrimps. In view of the contentious issues surrounding shrimp aquaculture, qualitative Life Cycle Assessment (LCA) has been applied to evaluate the principal environmental interventions over the life cycle shrimp production chain. Key issues have been identified and potential impacts from different farming systems were assessed and compared based on the qualitative LCA results and knowledge of systems. However, practical approaches permitting the quantification of environmental impacts associated with shrimp aquaculture such as depletion of wild broodstock, depletion of marine fish stock, loss of biodiversity, impacts on land use and impacts of chemical use need to be further investigated.

Keywords: *shrimp aquaculture; Life Cycle Assessment (LCA); Thailand*

Introduction

Thailand has been a major world shrimp producer since the 1990s. Export of farm-raised black tiger prawn (*Penaeus monodon*) products has benefited the whole country. Shrimp production has also generated substantial social welfare, particularly to people living in coastal zones. In addition, *u*pstream and downstream industries associated with shrimp production have generated numerous job opportunities particularly for women. However, the shrimp industry in Thailand has been controversial because of its associated potential environmental impacts. Several innovative culturing technologies has been developed and applied to minimise the negative environmental consequences especially at the farm stage which is believed to be the most important stage driving the whole industry. However, the wider environmental sustainability implications of these culturing practices have not been thoroughly demonstrated. In this study, qualitative Life Cycle Assessment (LCA) is applied to identify and assess the environmental interventions at different stages in the shrimp production chain to give a comprehensive view of their environmental consequences.

Shrimp Aquaculture in Thailand

Shrimp farming in Thailand originally developed as a by-product from the salt fields along the seacoast of the Inner Gulf of Thailand. The conversion of salt fields into shrimp ponds was fortuitous as the salt fields were close to the sea, thus offering ideal conditions to obtain the wild shrimp larvae and facilitating shrimp farming. Several factors have also promoted the rapid growth of shrimp farming: the regional decrease of shrimp from wild capture, increasing demand for farmed shrimp, and a massive collapse of shrimp farming due to disease and pollution in Taiwan. Subsequently, the growth of the shrimp market has been the main driving force promoting rapid expansion of the shrimp farming in Thailand. At the beginning, shrimp ponds were constructed close to, or within, the mangrove area to obtain wild juvenile shrimps and to facilitate pumping of seawater into and out of the farms. However, the unfavourable conditions of acidic water due to high pyrite content and pond bottoms made uneven by roots led to the abandonment of numerous farms. The failure of shrimp ponds in mangrove areas initiated shifts in development of new ponds to more suitable and productive sites in coastal areas. This included the development of inland shrimp farming at low salinity in the Central Plain (Suphanburi, Nakhon Pathom, Ayuthaya, Angthong, Prachinburi, Nakhon Nayok and Chacheongsao). The emergence of inland shrimp farming in the most intensive and productive area of rice cultivation produced a heated debate over the environmental, social and economic implications of the activity. Nevertheless, it has finally been banned due to its potential impacts.

The main hubs of shrimp production presently are in the eastern and southern parts of the country. However, with uncontrolled shrimp pond expansion and inappropriate culture practices, shrimp farming in Thailand has been through ‘boom and burst’ cycles with very high production followed by a sudden collapse caused by outbreak of disease associated with poor water quality management (Nissapa and Boromthanarat, 2002). This has raised concern for the sustainability of shrimp aquaculture.

In addition to the concern over unsustainable farming practices, the social and economic sustainabilities of shrimp farming industry have also received much attention. As shrimp farming utilises common property resources, this tends to affect social equity distribution leading to competition for access to resources. From an economic perspective, even though shrimp farming is considered as a ‘quick-in-return’ investment, the financial performance of shrimp farming is unpredictable. Shrimp farmers have to invest a considerable amount of money for crop production due to the high capital investment of the shrimp ponds as well as the high cost of input factors. Moreover, production is subjected to climate variability, the possibility of disease outbreak and other adverse factors resulting in crop failure. The profitability of the crop is intimately associated with current shrimp price which are subject to the supply and demand of the global market. In light of these factors and the limited control that a farmer can have over them, shrimp farming, to a certain extent, is an inherently risky investment.

In view of the shrimp industry outlook, the intensive farming system is still a common practice. However, the farm expansion has been limited because of the less land availability and the previous experiences of boom-and-burst consequences. Fishmeal shortage has also become more serious and this is linked to the production of trash fish that are not being sustainably managed. Much is still unknown regarding disease prevention and control. Vaccination against diseases has not been successfully developed, and attempts to prevent or eradicate diseases have failed until now. Besid*es,* Thailand has been faced with the export problems due to the increased tariff rate by the European Union (EU) to revoke the Generalized System of Preferences (GSP) privileges since 1999. This has caused a decrease in Thai exports to the EU (Sitthipongpanich, 2001).

Non-tariff barriers, such as food safety requirements and certification and eco-labelling schemes, have also been proposed. The shrimp market is still uncertain, and the price fluctuates depending on the global demand.

Research Methodology

The study has focused on the shrimp aquaculture production in Thailand. Following a cradle-to-grave approach, the life cycle stage of shrimp production includes trawling (capture of wild broodstock), hatchery (post-larvae rearing), farming, processing, storage, transportation to importing countries, consumption and waste management. Qualitative LCA methodology has been used as the analytical tool to identify and assess the environmental impacts associated with shrimp production. Due mainly to concerns over the environmental sustainability at the farm level, five farming types with diverse culturing techniques and farming management systems were selected from the shrimp farms in the Eastern region. These five case studies, which have been analysed, are representative of current shrimp farming practices in Thailand. Features specific to the individual farming system are respectively: “biological and Code of Conduct (CoC)” farming applying the CoC (Code of Conduct for Responsible Marine Aquaculture, the environmental management programme of shrimp farming developed by Department of Fisheries, Thailand) as well as biological management strategies such as using mangrove to filter organic matter to minimise water pollution and minimising of chemical usage; “conventional and CoC” farming applying the concept of environmental management system to minimise the impacts and only approved drugs can be used when necessary; “probiotic” farming utilising probiotic microbes to digest waste in order to maintain the water quality during culturing; “ecological” farming aspiring to raise shrimps naturally by optimizing input factors in order to sustain the pond productivity with no use of chemicals; and “ongoing-to-be organic” farming operating at lower stock density (not more than 312,500 of post-larvae per hectare) with best available organic inputs and chemical usage is completely eliminated.

Environmental interventions in every stage along the block-frozen shrimp production chains were qualitatively identified. Site visits to hatcheries, farms, and processing plants were conducted to understand fully the nature of their activities. Hatchery operators and shrimp farm managers were interviewed to obtain more information on hatchery operations and farming managements. Processing lines at the shrimp processing factory were “walked through” to ensure familiarity with the processing method used. Information regarding the transportation of product to destinations by refrigerated container ship was obtained by interviewing the shipping company.

Life Cycle Stages of Shrimp Production

Shrimp culturing starts with the capture of wild broodstock so that the hatchery cycle can be established for producing post-larvae. At the hatchery, the broodstock is subsequently induced to spawn through photoperiod manipulation. After hatching, a combination of phytoplankton, artemia cyst and artificial diet is fed until the hatchlings transform to post-larvae stage 15, where they are transferred to culturing ponds. At the farm, the ponds are constructed with a suitable depth and bank slope and their bottoms are compacted to provide an appropriate habitat for shrimps. Water is introduced into the pond and the water quality is prepared into an optimal regime for shrimps. After that, the stocking is carried out with a prior acclimatization of post-larvae. High-protein artificial feed is used for growing shrimps. During culturing, water exchange is carried out in order to dilute the waste accumulated in the pond. Aeration systems are used to maintain a sufficient oxygen level in the ponds and to allow organic decomposition to take place. The crop production takes about 120 days or more. At harvesting, the water is drained out of the pond through a net. Shrimps are collected and immediately put into icy waters. Thereafter the sizing is performed and they are subsequently transported by refrigerated truck to Central Auction Shrimp Market, where the shrimps are further distributed to processing plants. The shrimps are processed into block-frozen shrimps and transported to overseas by refrigerated container ship. At post-harvest, accumulated sludge is taken out from the pond bottom for further treatment and disposal. The soil surface is turned over and the ponds are allowed to dry out before a new crop is introduced.

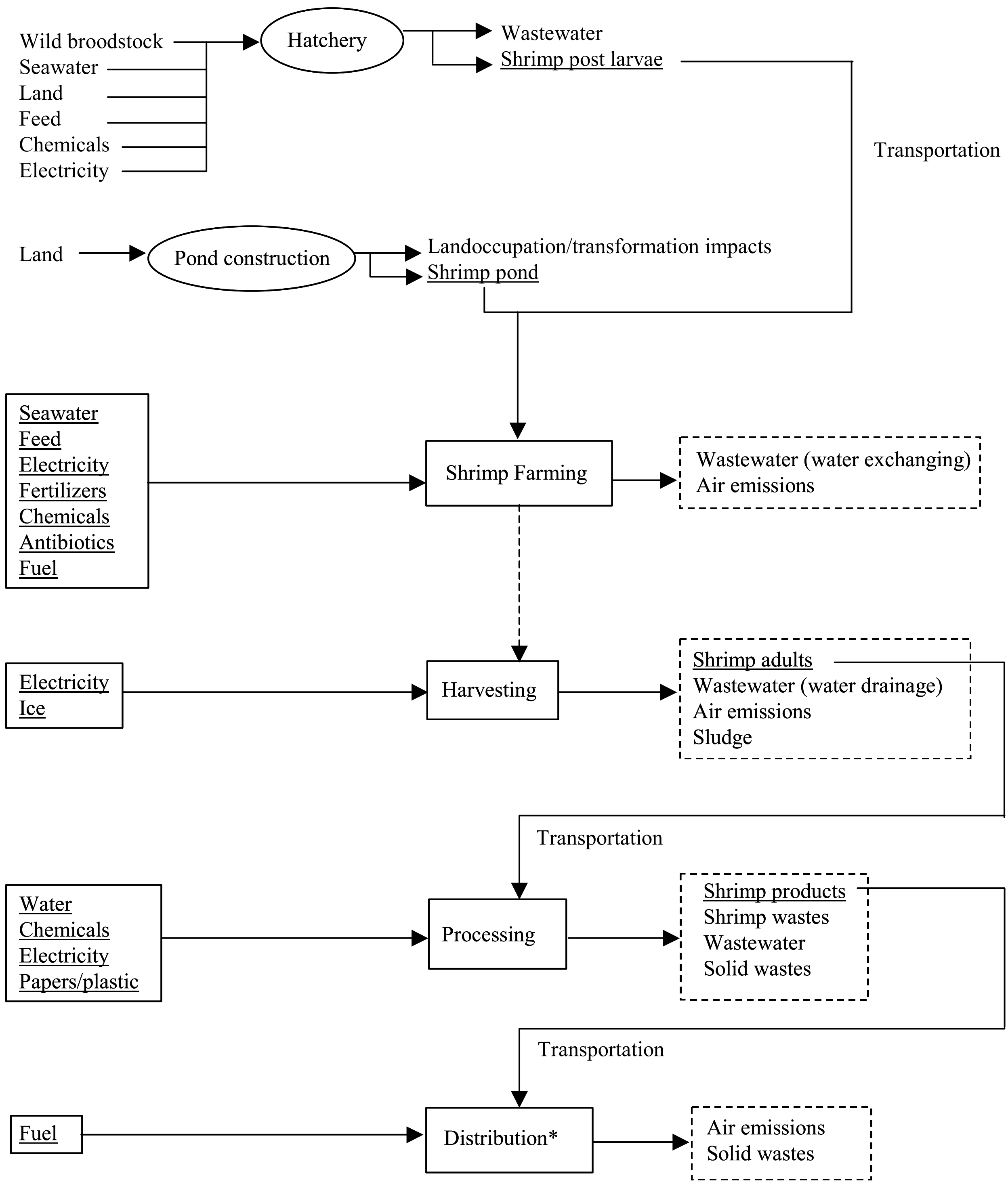
Results and Discussions

The principal life cycle stages of shrimp production are shown in Figure 1. Through the qualitative LCA analysis, the environmental impacts associated with shrimp aquaculture production chains have been identified, as follows. Trawling for wild broodstock has a potential to cause a stress on marine ecosystems such as destruction of seabed, changes of benthic communities, and loss of biodiversity. Introduction of the wild broodstock and post- larvae can also be a source of pathogens or diseases. At the hatchery, artemia cyst and artificial feed are the most crucial inputs in addition to the introduction of broodstock; of these the utilization of resources for producing feed and use of energy for harvesting and processing artemia cysts are likely to have the most significant impacts. Intensive farming system required higher stocking rate and this might lead to the deletion of wild shrimp populations.

At the farm site, the change of land use as a result of the construction of shrimp ponds could affect the soil quality and biodiversity. Land requirement of shrimp ponds may come into competition with other possible land uses when the scale of activities increases. Pond management, during and after the culturing period, highly influences the degree of water and soil quality deterioration. The higher demand of trash fish to supply fishmeal as well as fish oils required for formulated shrimp feed production may possibly increase pressure on the capture fisheries leading to the depletion of wild catch as well as marine fish stock. The over-harvesting of juvenile fish and non-target species might affect marine food chains and consequently marine biodiversity and ecosystems. Uneaten feed as a result of poor feeding management is the main source of nutrients dissolved in wastewater and sludge accumulated in shrimp ponds. However, using microbes to digest protein wastes together with sufficient aeration is likely to reduce air emissions such as hydrogen sulphide and ammonia which are highly odorous and detrimental to shrimps' health.

Another aspect at the farm stage is the ecological toxicity of chemicals and antibiotics used. Excessive use of chemicals and antibiotics possibly create the problem of antibiotic resistance development and residue accumulation, and toxic effects on non-target species in the ponds and surrounding ecosystems may change the ecological structure. Accumulation of the chemical used for improving soil and water quality as well as for treating diseases during the culture is a potential problem. Nevertheless, the human activities, such as removing sediment from the bottom pond, adding fertilizer or lime to improving the soil quality, and drying the pond between crops, influence the transport and fate of pollutant chemicals. At post-harvest, improper discharge of wastewater and sludge could affect the water supply quality apart from the external environmental quality. Moreover, some diseases could directly be triggered or spread more effectively by poor environmental conditions. Human toxicity associated with antibiotic contaminated shrimps is probably not an issue as potentially hazard antibiotics and chemicals have been banned, and thereby eliminated from shrimp farming in Thailand. Water and electricity consumption are the key issues at a processing plant. Air emissions from the fuel used by shipping vessel as well as the energy consumed by refrigerated containers are the most significant environmental impact from the transportation stage. Food waste generated i.e. shrimp shells should be disposed properly.

Focusing on the farm stage, the environmental impacts of different shrimp farming systems were compared based on the results from the qualitative LCA and the knowledge of farming systems. Conventional and CoC farming types used a considerable amount of seawater due to the higher rate of water exchange during the culturing period. The energy used for aeration systems in this farming type was also fairly high. The wastewater treatment using mangrove wetland systems at Biological and CoC farm seemed to be an effective mean to reduce the organic loads of discharge. The probiotic manufacturer has claimed that the probiotic products have been proved that they are environmentally safe and sound. However, the mode of action of probitoic substances associated with their remediation effects was not clear. Inputs of probiotic farming were not significantly different from conventional intensive farming, except the probitoic substances. Ecological farming consumed fewer resources because of no water exchange during the culturing period and the under-rate feeding management. Organic farming used less energy due to the lower stocking rate. Considering the key factors affecting the environmental performance of shrimp farms, two main issues identified are farm location and choice of culturing practice. The comparison of the impacts from different shrimp farming systems is not straightforward if the farms are located in the different geographical areas.



*Distribution means the distribution from storage to wholesalers, retailers and consumers

Figure 1. Principal flow chart of the life cycle of shrimp production.

This is due to the intrinsic properties of geographical location as well as climate conditions attached to a particular site. In searching for the most environmental preferable farming system, it is rather sensible to apply a qualitative LCA to assess the potential environmental impacts associated with the choices of site location and culturing technique during the planning phase of site selection and for pond management during the farming operation. Another issue of concern related to the comparison at national level, between countries, is that there are significant differences between and within countries regarding the levels of production intensity, types of resource utilisation, farm numbers and their sizes (Kongkeo, 1997). Most shrimp farms in developing countries are operated by a family. How to aggregate the assessment of individual farms to assess cumulative effects into an “integrated” local, or even national, impact assessment are still problematic. Some effects and impacts are still unknown, such as fate and transport of chemicals used previously, and impacts on biodiversity. Other barriers to LCA studies, in Thailand as well as other developing countries, are a lack of LCA expertise and database systems. From the qualitative LCA results, nevertheless, careful site selection can help minimise unnecessary impacts such as selecting areas without flooding, avoiding areas with potential sources of contamination and to make sure that the selected area have enough available sources of water supply. The choice of farming system, especially the intensity of inputs and management strategies, should be selected based on the particular environmental condition of the site. For instance, the stocking density should not exceed the pond’s carrying capacity that is associated with the pond soil and water properties to accommodate the shrimps’ growth. Even permitted chemicals should be used as little as possible and only as necessary. Primary treatment of wastewater drained from the shrimp pond after crop production is needed, for example by sedimentation. Growing fish species in alternate periods, for instance Tilapia, could reduce the nutrient level in the wastewater. Accumulated sludge in the pond bottom soil has to be managed properly, for example by collection in a sludge storage pond for further disposal. Pond drying between crops should be long enough to allow the decomposition process to come to completion. Transportation time from farm to table should be minimised and efficient use of water and energy in processing should be applied. Storage time at the processing plant before transporting to overseas should be as short as possible.

Conclusion

The environmental impacts associated with shrimp aquaculture are dependent on the characteristics of the site and the choice of culturing practice. The qualitative LCA results provided a better view of the environmental interventions over the life cycle of shrimp production. However, practical approaches permitting the quantification of environmental impacts associated with aquaculture need to be further investigated. Application of LCA studies of different farming systems and sites will ascertain the practices with the least environmental impacts to make the shrimp industry more sustainable.

References

Kongkeo, H. 1997. Comparison of intensive shrimp farming systems in Indonesia, Philippines, Taiwan and Thailand. *Aquaucture Research* 28: 789-796.

Nissapa, A. and Boromthanarat, S. 2002. A Case Study on Institutional Aspects of Shrimp Aquaculture in Thailand. Report prepared under the World Bank, NACA, WWF and FAO Consortium Program on Shrimp Farming and the Environment. Work in Progress for Public Discussion. Published by the Consortium. 72 pp.

Sitthipongpanich, T., 2001. The Thai frozen shrimp industry. *Industry Note: Food & Agribusiness Research.* Robobank International, the Global Food & Agribusiness Knowledge Bank: 1-5.

Life Cycle Assessment of trout farming in France: a farm level approach

Papatryphon, E.[*1], *Petit, J.*[1], *Van der Werf, H. M. G.*[1] *and Kaushik, S. J.*[2]

[1]*Environmental Systems Analysis Research Group, UMR Sol Agronomie Spatialisation, Institute National de la Recherche Agronomique, 65 rue de Saint Brieuc-CS 84215, 35042 Rennes Cedex, France.*

[2]*Fish Nutrition Research Laboratory, Unité Mixte INRA-IFREMER, Unité d'Hydrobiologie, 64310 Saint-Pée-sur-Nivelle, France.*

Abstract

The aim of this study was to evaluate the applicability of the Life Cycle Assessment (LCA) methodology for the comparison of the environmental impacts of individual farms and as a tool towards understanding and, therefore, assessing the effects of management choices/decisions at the farm level. Eight trout farms were used to obtain data for the inventory regarding the process of fish production. Results indicate that there is a significant variability in the environmental impacts among the farms and that most of the variability can be explained by a detailed analysis of farm parameters. Our results point towards the particular importance of two parameters-metrics, which could be used as indicators of overall environmental performance: 1) the feed efficiency and 2) the production intensity during the dry season, expressed as production per volume of fresh water use. We believe that LCA has good potential for the comparison of farming systems at the farm level.

Keywords: Aquaculture, rainbow trout farming, LCA methodology, farm systems analysis.

Introduction

The environmental impacts of aquaculture activities have received considerable attention, especially in terms of the production of carnivorous fish, like the salmonids (Naylor et al., 1998; 2000). Rainbow trout comprises approximately 68% by weight of farmed fish production in France, with an annual production of approximately 40000 tonnes, which sets it as the leading economic activity of the sector (FAO, 2001). Under the umbrella of the theme "Sustainable Aquaculture", the French National Institute of Agronomic Research and the French Inter-Professional Committee of Aquaculture Products have been engaged in the development and application of LCA for the evaluation of the environmental impacts of the various aquaculture sectors, but also as a tool towards optimum farm management and strategic planning.

LCAs in agriculture have been mainly conducted as a means to photograph the environmental impacts of production processes, and to further position them in a global context or to compare production processes widely different from each other. There have only been a few attempts, to our knowledge, aiming at using LCA on an individual farm basis, and which could serve as a tool to optimise farm management decisions and strategic planning (Haas et al., 2000).

[*] Present address: European Commission, Institute for Prospective Technological Studies, Edificio Expo, C/Inca Garcilaso s/n, E-41092 Seville – Spain, tel: +34 95 4488 402, fax: +34 95 4488 326, ilias.papatryfon@cec.eu.int

Goal and Scope

Aims and functional unit

The overall aim of the research project is to develop and apply the LCA methodology for the evaluation of the environmental impacts of fish farming in France, as a means to support the communication and decision making of the stakeholders involved, namely the fish farming industry, the research community, as well as the administrators responsible for putting forward the directives concerning the evaluation and evolution of the industry. The specific goal of the study described herein was to assess the potential of using LCA as a tool for the identification and demonstration of the potential variability in the environmental impacts due to different choices in farm management, such as in technical sophistication, product type, production planning, and geographical origin, to name a few. Since the trout production industry comprises the most significant aquaculture activity in France, it was selected as the first candidate for evaluation. At first stage, the production process is followed to the farm gate, the functional unit being one ton of trout live weight.

System boundaries

The system to be studied covers the farm production of trout, as well as the following upstream processes as related to non-renewable energy: a) the production and use of primary inputs to the farm (oxygen, veterinary treatments and feed), b) the production and transformation of feed ingredients, including the agricultural and fishery phase of the various ingredients, c) the production of equipment used on the farm, d) the construction and production of necessary infrastructure and e) transportation in all stages. For the assessment at the farm level, the production of the fish inputs as eggs or juveniles was not taken into account, so as to calculate and compare the efficiencies of each farm. Fish inputs were included however in the calculations of the final scenarios.

System description

Rainbow trout aquaculture in France is carried out in freshwater, flow-through raceway-type installations, mainly using river water, and following the intensive aquaculture model, as they rely completely on external provisions of feed. The most recent survey on aquaculture production was published in 2000 (Agreste, 2000), indicating the existence of three main product types (as defined by their size and not by other qualitative traits), namely: portion trout (250-300 g mean fish weight), large trout (900-1500 g mean fish weight) and very large trout (2000-3000 g mean fish weight). Approximately 50% of the trout produced is of the portion size while the other 50% is trout of larger sizes. From a geographical point of view, 50% of the production is carried out in only two regions, Aquitaine and Bretagne, on an approximately equal basis. Eighty percent of the production by weight is produced by only 20% of the production facilities, indicating the importance of large production capacity farms, also being the ones with the highest investments towards intensification, mainly related to water treatment (aeration, oxygenation, recycling etc). Data, concerning the year 2002, from eight individual farms that were located in Aquitaine and Bretagne, were collected in an effort to cover a large part of the variability regarding on-farm production practices, namely final product size and production intensity (expressed as fish production per unit volume of fresh water use during the driest month). The reason for choosing these two parameters as our basis of farm categorisation arises from our belief that on-farm production practises vary widely as these parameters change. The individual farms were also chosen based on their potential to be included in the different production scenarios (to complete the production cycle, from the broodstock to the final product), as well as the availability of data, and their willingness to collaborate and share data.

Production scenarios were constructed based on data obtained from the above mentioned farms in order to simulate as best as possible the complete production process for each individual farm. To this end, three separate production processes (corresponding to separate farm units) were used: A) The production of broodstock and eggs, B) the production of juveniles and portion trout and C) the production of large and very large trout. Two scenario types, which are also the most important production scenarios of the French trout farming, were identified in the farms we assessed: Type I, from the egg to portion size trout and Type II, from the egg to large and very large trout.

Allocation

Economic allocation was used for all processes yielding by- and co- products during the production of feed ingredients. On the fish farm, trout was considered as the only product and, therefore, allocation was not necessary. In farms where trout of various market sizes is produced, the product was considered to be identical, and a weighted average was used for categorisation. The production of manure from trout farms was considered as a waste product and as a result no allocation was used either.

Selection of impact categories

The impact categories that were included in this study are Eutrophication, Acidification, Climate change, Energy Use and Biotic Resource Use (expressed as Net Primary Production Use). The first four were chosen as they are among those that are considered critical for inclusion in all LCA studies (Guinée, 2002). Furthermore, Eutrophication, Energy Use and Biotic Resource Use were judged to be the most relevant categories for the system we studied, and extra care was taken towards their assessment. Biotic Resource Use was calculated as the direct use of net primary production, while indirect use, such as that due to land use or other activities, was not taken into account (Papatryphon et al., in press).

Inventory Analysis

Data regarding inputs and outputs to, and from, the farms, were directly obtained from producers records. Information regarding the data and assumptions for the production of feed can be found elsewhere (Papatryphon et al., in press). Data on oxygen production and transport, as well as the production of equipment used on the farms were obtained from experts. Data regarding the infrastructures were calculated based on data obtained from each farm. Emissions regarding all agricultural phases were calculated according to Papatryphon et al. (in press).

Results

Environmental Impact Assessment – farm level

Table 1 shows the range as well as the relative variation for the impacts associated with the five impact categories included in this analysis. The assessment at this stage does not include the upstream processes regarding the production of fish that supply the farm, since we are interested at obtaining the impacts at the farm level (and not the production scenarios). A variation of 40-50% is found for most impact categories except for energy use, in which case the variation reaches approximately 90% and for biotic resource use, for which it is lower at 24%. In an effort to identify the factors responsible for the variability in the calculated impacts, the farms were categorised in three groups relative to the final product type (Table 1), indicating that there is an increasing trend for all impact categories as the size of the final product increases.

Table 1. Variation in the total calculated impacts among seven individual farms categorised per final product type (as main product) for the production of 1 ton of rainbow trout live weight.

		Portion trout	Large trout	Very large trout	Relative variation (%)[1]
Number of farms		2	3	2	
Eutrophication	kg PO_4-eq	46.3-63.8	58-72	65.9-74.8	47
Climate change	kg CO_2-eq	1760-1850	1960-2290	2430-2760	44
Acidification	kg SO_2-eq	12.1-13.7	14.4-17	16.5-19.1	45
Energy use	MJ	30000-42300	41000-57900	51900-78200	89
Biotic resource use	kg C	48700-50700	53000-57400	60500-62200	24

[1] Relative variation = (maximum value - minimum value)/((minimum value + maximum value)/2)

Contribution analysis – farm level

A process contribution analysis was conducted for all relevant impact categories. The analysis indicates that feed production has the highest contribution to the impact categories biotic resource use (100%), climate change (mean 83%, range 77-88%), acidification (mean 82%, range 75-87%) and energy use (mean 52%, range 36-73%), while it is responsible for only 6% (range 5.5-6.6%) of the impact towards eutrophication. Fish production is the major contributor to eutrophication (mean 94%, range 93-94%). Electricity and liquid oxygen use are the second and third most important contributors to energy use, respectively, followed by infrastructure, equipment, diesel and formol production. In order to understand the variation within each product category, the farms were also categorised based on the production intensity. A comparison of the energy use among the different farms indicated, that there is a clear trend towards an increase in energy use with increasing production intensity and with increasing average product size and, therefore, these two parameters may be determinant factors for this category (Table 2). The variation and larger overlap in the impact to eutrophication between the different product types is mainly related to the variation in the feed efficiency encountered between the different farms and to the difference in the degree of solid waste removal (from 0 to approximately 30%).

Table 2. The variation in energy use for the production of 1 ton of rainbow trout live weight in 7 farms categorised by production intensity and final product size (as main product).

Category	Production intensity (g production/m^3 fresh water use)	Final product size	Number of farms	Energy Intensity (MJ)
1	low	Portion	1	30000
2	medium	Portion	1	42300
3	medium	Large	1	41000
4	medium	Very large	1	51900
5	high	Large	2	57600
6	high	Very large	1	78200

Environmental Impact Assessment - production scenarios

Table 3 shows the range as well as the relative variation for the impacts associated with the five impact categories included in the analysis of the two production scenarios. The relative variation is similar from what was found in Table 1 in the comparison at the farm level, since in the production scenarios the final impact is higher but it increases more or less equally for both scenarios.

Table 3. Variation in the total calculated impacts among two farm production scenarios (I: portion trout and II: large and very large trout) for the production of 1 ton of rainbow trout live weight.

		Scenarios		Relative variation (%)[1]
		I	II	
Number of farms		2	5	
Eutrophication	kg PO_4-eq	48.4-65.4	68.8-79.6	49
Climate change	kg CO_2-eq	1800-1902	2120-2877	46
Acidification	kg SO_2-eq	12.4-18.7	16-20.3	48
Energy use	MJ	31517-44166	44664-80884	88
Biotic resource use	kg C	49662-51962	59797-65753	28

[1] Relative variation = (maximum value - minimum value)/((minimum value + maximum value)/2)

Contribution analysis – production scenarios

Table 4 shows the contribution of each production process to the impacts of the complete production scenarios. For simplicity and since the relative contributions of each production process (A, B or C) to the final impacts were similar for all impact categories, a range is presented. For Scenario type I, the process concerning the production of the final product has, by far, the largest contribution (95-98%). Similarly, the production process of the final product is also the most important contributor for the Scenario type II (78-97%), although the production of juveniles/ portion sized fish can become an important contributor to the overall production process as well (3-21%).

Table 4. Contribution of individual processes to the final impacts for the two production scenario types studied (range of % contribution for the different impact categories and different farms used).

Processes included	Scenario Type I Range	Scenario Type II Range
A. Production of large and very large trout		78-96.7
B. Production of juveniles and portion trout	95.2-98	3.2-21
C. Production of broodstock and eggs	2-4.8	0.07-1

Discussion

Farm impact assessment

A comparison among the different farms indicates that there is significant variability in all environmental impacts for the same product species, namely the rainbow trout. As the rainbow trout industry is the most mature and competitive aquaculture industry in France, it has evolved in a way that it has diversified in terms of the production practises. The most important difference regarding the production orientation among trout farms is the final product size which, by default, has significant implications in the zootechnical, environmental and economic performance of the farms. Results from this analysis, which is based however on a limited set of farms, indicate that as final product size increases so do the environmental impacts for all impact categories assessed, when expressed on a unit mass production basis. For the impact categories Biotic resource use, Climate change and Acidification, feed is the major contributor to the environmental impacts. Furthermore, feed efficiency is directly related to the size of the fish being produced, since as fish size increases, feed efficiency decreases. It is, therefore, not surprising to find that there is a correlation between the calculated environmental impacts and the final product type, which is due, in turn, to the underlying correlation of fish size and feed efficiency. A more detailed analysis regarding the environmental impacts of feeds using LCA has previously been conducted (Papatryphon et al., in press). For the impact category Eutrophication, the process of fish production is by far the greatest contributor. However, since the vast majority of eutrophying potential on the farm is of feed origin, it is clear that the variation in feed efficiency will also be the major parameter affecting the score to this category. Solid waste removal in French farms is not widely practiced, and from the farms assessed, only four had in place some type of removal. Two types of waste removal are mainly used in French fish farms, either filters specially designed for this purpose or the use of sedimentation basins at the outlet of the farms. When the removal of solid waste is not taken into account, there is a clear trend towards an increase in the impact to eutrophication as final product size increases. However, due to the existence of the removal technologies on some of the farms, the reported overlap in eutrophication was observed. For the impact category of energy use, the situation seems to be a little different, since a large part of the impact is due to the use of electricity on the farms, but also to the use of other high energy carrying inputs, such as liquid oxygen. Since electricity and oxygen are mostly required when the production intensity is high, which in turn may be due to farm management practices (elevated stock size, high feed distribution) and to environmental parameters (low river flow, high temperature) a good correlation was observed between the production intensity during the dry month and energy use. A correlation was also observed between final product size and energy use, as feed is also an important contributor to energy use and as feed efficiency varies with fish size. It seems, therefore, that feed efficiency and production intensity are the deciding factors affecting energy use. Furthermore, as high production intensity is related to high level of fuel and electricity use (direct or indirect, i.e. through oxygen use), it follows that feed efficiency and production intensity are also the deciding factors affecting the impact categories climate change and acidification.

Farm Scenario impact assessment

An assessment of complete production scenarios is necessary from an LCA point of view, in order to obtain a complete picture of the life cycle impacts of a product. Our assessment indicated that the large majority of the potential impacts is associated with the production process found at the end of the production chain, as would be expected, since very small quantities of the processes located upstream are required for the production of the final product. Moreover, the effect of the fish hatchery on the overall impact is very small, which is due to the high fecundity of fish, as compared to other farm animals. Nevertheless, it is of interest that large variations can be found in the contribution of the production of juveniles and portion trout to the production of large and very large trout, which indicates that the overall environmental performance of a product may be more or less different from the environmental performance of the last stage of production. We believe that the variation calculated herein may provide a good indication of the expected variability of the potential environmental impacts of French trout farming under the impact assessment methodology used.

References

Agreste 2000. *Recensement de la Pisciculture Française en 1998 : Salmoniculture et Pisciculture Marine.* Agreste Données Chiffrées Agriculture, No 124. Ministère de l'agriculture et de la pêche, Service Central des Enquêtes et Etudes Statistiques, Paris, 128 pp. (In French).

Guinée, J.B. 2002. *Handbook on Life Cycle Assessment. Operational Guide to the ISO Standards*, Kluwer Academic Publishers, Dordrecht, 708 pp.

FAO, 2001. FAO yearbook, Fishery statistics, Aquaculture production 2001. Vol 92/2.

Haas, G., Wetterich, F. and Geier, U. 2000. Life Cycle Assessment framework in agriculture on the farm level. *Int. J. LCA* 5, 345-348.

Naylor, R.L., Goldburg, R.J., Mooney, H., Beveridge, M., Clay, J., Folke, C., Kautsky, N., Lubchenco, J., Primavera, J. and Williams, M. 1998. Nature's subsidies to shrimp and salmon farming. *Science* **282**, 883-884.

Naylor, R.L., Goldburg, R.J., Primavera, J.H., Kautsky, N., Beveridge, M.C.M., Clay, J., Folke, C., Lubchenco, J., Mooney, H. and Troell, M. 2000. Effect of aquaculture on world fish supplies. *Nature* **405**, 1017-1024.

Papatryphon, E., Petit, J., Kaushik, S. J. and Van der Werf, H. M. G. Environmental impact assessment of salmonid feeds using Life Cycle Assessment. Ambio, in press.

Environmental impacts from Danish Fish Products

Mikkel Thrane
Department of Development and Planning. Aalborg University,
Fibigerstraede 13, 9220 Aalborg, Denmark. Phone +45 96 35 83 16, e-mail thrane@i4.auc.dk.

This article presents the results from an environmental assessment of Danish fish products carried out as part of a PhD project at Aalborg University. The article includes data for energy and water consumption for a wide range of fish products, but also presents the results of a life cycle assessment (LCA) of 1 kg frozen flatfish filet. It is concluded that the fishing stage is one of the most important stages, not only for flatfish – but Danish fish products en general. It is argued that the fuel consumption represents an important impact potential and that improvements in fuel efficiency are consistent with other objectives, such as a reduction of discard, overexploitation and seabed impact. In this regard, it is shown that that the fuel consumption can be reduced with up till a factor 15 in some cases. It is suggested that more attention should be given to the development of cleaner production methods in the fishing stage, and that the previous focus on the processing stage in terms of environmental regulation and cleaner production can be characterized as sub-optimization from a life cycle perspective.

Key words: Danish fishery, Energy consumption, LCA, Flatfish, Environmental policies

Introduction

As environmental policy becomes more product-oriented, an increasing need for seeing environmental impacts in a life cycle perspective (from sea to table) is occurring. Traditional environmental regulation mainly focuses on the companies and their (on site) emissions, but frequently larger environmental impacts are found elsewhere in the life cycle.

So, far few studies have been made of the environmental impacts of fish products from a "cradle to grave" perspective, but recently LCA studies from Sweden and Island have been published suggesting that the fishing stage is the environmental hot-spot for Atlantic cod (Eyjólfsdóttir et al., 2003; Ziegler et al. 2003). Previously, Danish LCA screenings of pickled herring, canned mackerel, and frozen peeled blue mussels have come to the same conclusion (Christensen et al. 2001).

The studies mentioned above only include a limited number of fish species and have some methodological shortcomings in relation to co-product allocation (typically handled by mass or socio-economic criteria) and representativeness – especially in previous Danish LCA screenings.

The dissertation "Environmental Impacts from Danish Fish Products" (Thrane, 2004a) includes a detailed analysis of the environmental impacts from Danish fish- and shellfish products. The analysis represent state-of-the-art LCA methodology (that is a consequential approach), which is characterized by market based modelling of the product system, use of marginal data and the application of system expansion for handling of co-product allocation (Ekvall and Weidema, 2004). The methodological aspects are further described in the following.

Methodological aspects

The environmental assessment in Thrane (2004a) is carried out in three separate steps: a MECO analysis, a quantitative LCA, and finally a qualitative LCA:

- The MECO analysis provides information about the inputs and outputs of materials, energy, chemicals, and other aspects - for all major species groups, and can be perceived as an LCA without the impact assessment phase.
- The LCA includes a detailed LCA of flatfish but LCA screenings are also made of other fish products for the purpose of generalization.
- Finally, the qualitative LCA provides an overall assessment of impact categories, which have not been included in the previous LCA, addressing all major species groups.

The present article presents key findings from the MECO analysis, as well as the results from the detailed LCA of flatfish - but results for the LCA screenings and the qualitative LCA are also discussed briefly.

Data collection

Data for the fishing stage has been relatively detailed. The data include a sample of 330 vessels representing a population of 1528 vessel making up 99% of the total Danish catches, measured in standard-catch-value. (Danish Research Institute of Food Economics, 2001). These data have been supplemented with data for specific fishing gear and vessels sizes (Nielsen, 2002) and personal interviews with fishermen, slip-ways and producers of antifouling agents.

The data for fish processing are based on literature studies supplemented with empirical studies. The data for the processing stage represent 10 different plants and the main references are Andersen et al (1996), Matcon A/S, and Dansk Energi Analyse A/S (1995). In the LCA of flatfish, detailed empirical data from one large plant have been applied as the main source.

For the wholesale- and transport stage, I have used a combination of empirical data (one company in each case) and theoretical calculations based on literature. For the consumer stage, I have only used literature data, while the data for the retail stage are based on empirical data from three supermarket chains combined with literature data for cooling processes etc. For more details, see Thrane (2004a)

Co-product allocation
Both for the detailed LCA of flatfish and the LCA screenings of other fish products, co-production allocation has been avoided through system expansion in the fishing stage, where several species typically are caught at the same time (target fish and by-catch). System expansion has also been applied at the processing stage, where a number of by-products occur after the filleting process. In both cases, the applied methodology follows the principles for co-product allocation described in Weidema (2001). Allocation by physical causality (weight or volume) has been applied for most other stages / processes, while economical allocations have been used for exchanges related to shopping at the use stage.

Data presented for energy consumption in this article are slightly overestimated, especially for the processing stage, as avoided exchanges haven't been considered in the MECO analysis.

System delimitation
Material flows representing more than 0,1% of one kg of fish product dispatched from the processing stage are included, but for certain chemicals such as anti fouling agents, I have included smaller flows. The cut-off criteria for energy consumption have been 0,1 MJ per kg of frozen flatfish, dispatched form the processing stage.

As mentioned, market based system delimitation has been applied. As the purpose of the study has been to elucidate the exchanges and impacts from Danish fish products in a regulation perspective, the product chain has been fixed to Danish producers and European supermarkets and consumers. The latter is because more than 90% of the Danish fish most are exported – mainly to the European market. Hence, compared to an attributional LCA, the consequential approach has not affected the system delimitation with respect to the immediate product chain (foreground data).

Other methodological aspects of the LCA
In the LCA case study of flatfish the functional unit is one kg consumed frozen flatfish filet (IQF) in consumer packaging made of cardboard in boxes of 300 gram each. The results presented here only include characterization (the first step of impact assessment). An updated version of the method known as EDIP 96 described in Wenzel et al. (1997) has been used.

The PC tool "SimaPro version 5.1" has been used for the calculations and the updated EDIP method is entered by 2.-0 Consultants as part of the LCAfood project. Several databases have been used for related processes, such as materials, chemicals, and energy. In this regard, I have used the ETH-ESU database, which includes capital goods for all energy processes. The database is around 10 years old and large uncertainties exist. Specifically for the direct emissions at the fishing stage, I have used data from the European Environmental Agency (European Environmental Agency, 2001). Data from the Danish LCAfood project have been used for different types of food ancillaries and avoided emissions such as soy protein and minced pork (LCAfood, 2003). This project also uses the ETH-ESU 96 database and most of the data are also based on system expansion.

MECO analysis - exchanges in a life cycle perspective

The MECO study includes an analysis of exchanges for eight product chains. All exchanges are measured per kg consumed fish product and the exchanges have obviously been adjusted for product spillage along the product chain. The results for energy and water consumption are illustrated in the following.

Energy consumption (Heat and combustion)

The studies of the direct heat and combustion related energy consumption show that the fishery is the most important life cycle stages for demersal fish and shellfish (see table 1). Norway lobster represents the largest energy consumption (roughly 1000 MJ per kg consumed lobster meat) of which the fishing stage makes up 95%. For flatfish and other shellfish the energy consumption is well over 100 MJ per kg consumed fish. Other important life cycle stages are the use stage, but also processing for products that involve boiling such as shrimps.

Table 1. Energy consumption for heat and combustion in MJ per kg consumed fish product - for eight product types. All products are frozen except the pelagic species.

	Demersal fish (frozen filets)		**Shell fish (boiled, peeled & frozen)**				**Pelagic (pickled/canned)**	
	Codfish (block)	Flatfish (IQF)	Prawn (IQF)	Shrimp (IQF)	N. lobster (IQF)	Muss. (IQF)	Herring (in jar)	Mack. (canned)
Fishery	41	110	101	136	941	5	16	5
Landing	1	1	1	1	0	4	0	0
Processing	2	3	12	12	12	6	3	3
Wholesale	0	0	0	0	0	0	0	0
Transport	1	1	1	1	1	1	4	2
(if whole)	4	5	5	5	8	9	2	2
Retail	0	0	0	0	0	0	0	0
Consumer	14	11	16	16	44	12	9	9
Total	59	126	130	166	998	28	32	18

The energy analysis shows great differences between the energy consumption in the fishing stage, depending on the fish species that are caught. For instance, the energy consumption for Norway lobster is 6 litre per kg caught whole lobster, while it is 1 litre per kg caught whole flatfish, 0,36 litre per kg caught whole codfish and 0,06 litre pr kg caught whole mackerel (Thrane, 2004b) Energy consumption (electricity).

I have not included a separate table with information about the electricity consumption, but roughly 50% of the total electricity consumption is related to the use stage for all products. The total electricity consumption varies between 8 MJ and 28 MJ per kg. The consumption for the processing stage only represent 1-5 MJ per kg consumed fish products - depending on the species. Even though electricity consumption only represents around 40% of the electricity produced, it must be concluded that the heat and combustion related energy consumption is more important.

Water consumption

The most important stages with respect to ground water consumption are the processing- and use stage. The water consumption in the processing stage can be more than 100 litre per kg consumed shrimp and prawn (partly because of the boiling process), while it is around 10-30 litre for other species. For the use stage the water consumption is assessed to be around 15-20 litre per kg consumed fish - slightly more if dishwashing is done by hand and slightly less when dishwashing machine is used.

Considering the differences between different species, the picture is somewhat similar to energy consumption. Shellfish represents the largest exchanges (Norway lobster, prawn, shrimp and mussel), followed by demersal (flat- and codfish) and pelagic fish (herring and mackerel) that have relatively small exchanges at all stages.

LCA case study of flatfish

As mentioned, the MECO analysis does not include impact assessment. That is why the LCA is carried out. The characterization results for one kg consumed frozen flatfish filet (IQF) caught by the average fishing method, are illustrated in figure 1. Each impact category is separately discussed in the following. As it appears the dominating stages are fishing, use, and retail, but the various impact categories are further discussed in the following.

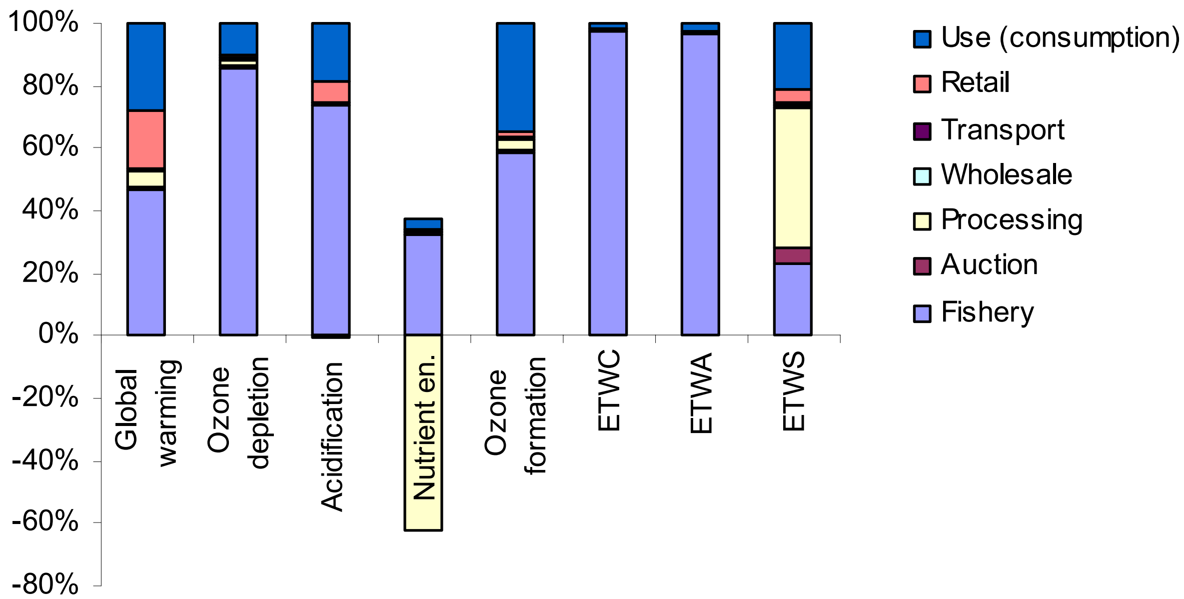


Figure 1. Characterization results for one kg consumed flatfish filets (IQF). Based on the Danish EDIP method.

GWP – gram CO_2 equivalents
As it appear in figure 1, the it is roughly half of the global warming potential (GWP) that is related to fishing, while the remaining mainly originates from the use and retail stage. Combustion of diesel is the dominating process for fishing while electricity consumption is dominating for the retail stage. For the use stage, it is partly electricity for cooling and food preparation, and partly combustion of fuel for transport, which are the important factors. The Dutch method ECOindicator 99 method shows similar results.

ODP - CFC_{11} equivalents
The potential contribution to stratospheric ozone depletion (ODP) is also mainly caused by fishing (~90%) due to the production of diesel, which involves Halon 1301, and the emissions of HCFC-22 cooling agents that are used on the fishing vessels. Again, the EcoIndicator 99 shows similar results. It is estimated that the uncertainty is somewhat larger than for global warming, as several of the ozone depleting substances occurs only because the ETH database is relatively old.

Acidification potential – gram SO_2 equivalents
For the acidification potential, the largest contribution also originates from fishing, mainly due to NO_x emissions. As for the GWP and ODP, the use and retail stage are also somewhat important, mainly due to SO_x emissions. When the EcoIndicator (99) method is applied, the fishery dominates even more.

Methodological aspects: According to Wenzel et al. (1997) contributions to acidification in robust areas such as the sea are less significant. As most of the Danish fishing activity takes place in the North Sea, it must be assumed that the actual acidification impact related to the fishing is significantly smaller than suggested above. Thus, it is probably more reasonable to argue that there are three potential hot spots: fishing, use and retail.

Eutrophication potential – gram NO_3 equivalents
As for previous impact categories, the fishing stage completely dominates for eutrophication, mainly due to NO_x emissions caused by the combustion of diesel. In contrast, the impact potential from processing is large but negative. The reason is that the by-products from processing are assumed to substitute soy protein, which is the marginal protein source on the global market according to Weidema (2003). The substitution of soy protein lead to avoided water borne emissions of nitrate in South America the overall contribution therefore becomes negative. The large avoided impact potential from South America hides that some emissions actually occur from the processing stage. Still the impact potential remains insignificant, as the emissions of wastewater are treated in modern biological wastewater treatment plants. When the EcoIndicator 99 method is applied the contribution from fishery becomes even more significant, but in this method N and P are not included in the characterization factors, and is therefore difficult to compare.

Methodological aspects: The eutrophication potential analyzed here does not distinguish between terrestrial and aquatic etutrophication, which is a weakness. Furthermore, site specific aspects are not included in the modelling, and the results are therefore quite uncertain.

Photochemical ozone formation potential – C_2H_4 equivalents
For photochemical ozone formation potential, the main contribution also comes from fishing followed by the use stage. In both cases, this is due to the emission of non methane VOC (hydrocarbons except methane), which is related to the production of diesel, but also the combustion process itself.

Methodology: This impact category shown in this paper does not distinguish between ozone leading to lower crop yields in agriculture, and ozone causing respiratory problems in urban areas. Nor are site-specific aspects included in the assessment. This is obviously a weakness. Substances that contribute to ozone formation may drift over long distances, but according to Wenzel et al. (1997) a site-specific factor between 0 and 1 should be applied for emissions in sparsely populated areas (low background NO_x levels) such as desserts or the sea. Hence, it would probably be more reasonable to argue that fishing and use both are potential hot-spots.

Eco-toxicity potential – measured in m^3 water or soil
The EDIP method includes three types of eco-toxicity: Eco-toxicity water (acute and chronic) as well as eco-toxicity soil (chronic). The unit is m^3 that express the water or soil volume required to dilute the given exchange to a concentration below the Predicted No Effect Concentration (PNEC). For further details see – Wenzel et al. (1997).

As illustrated in figure 1, both categories of eco-toxicity water are completely dominated by the fishing stages. This is caused by the emissions of TBT from the anti fouling paint. These results are plausible, as sex changes have been registered for 10 different kinds of Sea snails in Denmark, mainly because of biocides from antifouling agents such as TBT and copper compounds.

For eco-toxicity soil (chronic), the main contribution is the processing stage followed by the use-, auction- and retail stages. The most important substances are acetone followed by formaldehyde and hexane. This is also related to energy production. Normalized and weighted results are not presented in this paper, but it should mentioning that eco-toxicity soil (chronic) becomes completely insignificant after the normalization step. This shows that the processing stage remains relatively insignificant even tough the contribution to ETSC is large.

Future developments
Future scenarios suggest that the fishing stage will remain important even though TBT is substituted. The alternative antifouling pains are also somewhat problematic, and future scenarios suggest that fishery will remain dominating for eco-toxicity water (acute and chronic) even though the toxicity potential is greatly reduced from antifouling agents. Furthermore, the fuel

consumption is expected to grow in the fishing stage, while the energy efficiency probably will increase at other life cycle stages. For further details, see Thrane (2004a).

Other impact categories

There are a large number of other impact categories, which have been separately analyzed in a qualitative LCA – that is seabed impacts, land use, waste, use of non-renewable abiotic resources, use of groundwater, exploitation of fish, discard and by-catch, human toxicity potential, occupational health and safety, noise and accidents, and animal welfare.

The analysis shows that different conclusion can be obtained by focusing separately on other impact categories, but overall the qualitative LCA appear to strengthen the previous conclusions – thus pointing towards fishing as the overall hot-spot followed by use and retail. Fishing is characterized by overexploitation of certain fish stocks, a high frequency of injuries and accidents among fishermen (including fatal accidents), seabed impact inflicted by bottom dragged fishing gear (e.g. bottom- and beam trawl), by-catch of sea-mammals, and discard of fish.

Other fish products

Apart from the detailed LCA on flatfish, the dissertation includes a number of LCA screenings of other typical Danish fish products.

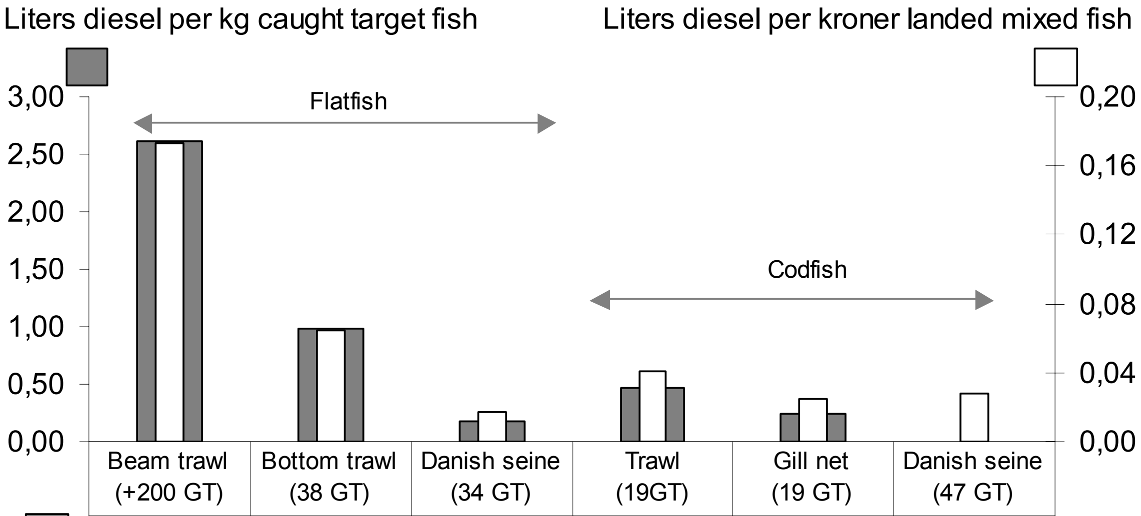
The screenings also suggest that the fishing-, use- and retail stages are the hot-spots (in this order) for products based on frozen shrimp, prawn and Norway lobster (all in cardboard packaging). Glass, aluminium or steel packaging would further increase the impact potential for processing, but these packaging materials have only been analysed for mackerel and herring.

For pelagic fish exemplified by pickled herring in glass jars and canned mackerel, the processing and transport stages are relatively more important, while the opposite is the case for the fishing- and retail stages. Still, the qualitative LCA points towards the fishing stage as the most important stage for all fish products. Thus, while the processing stage, can be important for some products due to packaging, the overall conclusion is that the hot-spots are indeed to be found in other stages of the life cycle – mainly the fishing and use stage.

Improvement potentials in the fishing stage

The analysis suggests that the largest potential for improvements is in the fishing stage. As it appear in figure 2, it is possible to reduce the fuel consumption per kg caught flatfish with a factor 15 by switching from beam trawl to Danish seine. If all flattish were caught by Danish seine or passive fishing methods such as gillnet, it would theoretically be possible to save around 30.000 m^3 fuel per year in the Danish fishery alone. This is 15% of the total fuel consumption in one year. This figure is rather theoretical and probably overestimated, but improvement potential also exists for other species groups.

Improvements can also be obtained by promoting the use of purse seine in the fisheries targeting herring and mackerel (and maybe other species), and by promoting the use of Danish seine, gillnet and long line in the fishery after codfish. The latter is also illustrated in figure 2. The negative fuel consumption for codfish caught by Danish seine is a result of system expansion – where the by catch substitutes relatively a more fuel consuming fishery.



	Beam trawl (+200 GT)	Bottom trawl (38 GT)	Danish seine (34 GT)	Trawl (19GT)	Gill net (19 GT)	Danish seine (47 GT)
■	2,61	0,98	0,18	0,46	0,24	-0,20
□	0,17	0,06	0,02	0,04	0,02	0,03

Figure 2. Fuel consumption per kg caught flat- and codfish in year 2000[1]. (Danish Research Institute of Food Economics, 2001; Nielsen, 2002; Beam trawler, 2001).

Figure 2 also shows the fuel consumption per landing value (white columns) for both flatfish and codfish. As it appear Danish seine is roughly 10 times more effective in this respects. For further details about potentials for improvements - see Thrane (2004a)

Conclusion

The MECO analysis as well as the LCA and the qualitative LCA points towards the fishing stage as the overall environmental hotspot followed by the use and retail stage for flatfish products. Still, LCA screenings of other fish products suggest that the hot-spot distribution is much similar for most other fish products, except for canned mackerel and pickled herring, where the processing and use stages are the hot spots. Still, if we include the results from the qualitative LCA, it is suggested that the fishing and use stage remains the overall spots for Danish fish products as such.

[1] The data are based on fishing vessel accounts. Flatfish (bottom trawl) is based on 16 accounts, flatfish (Danish Seine) comprises 9 accounts and flatfish (beam trawl) is based on interview with the owner of three trawlers. For codfish the number of accounts is 15 for bottom trawl, 22 for gill net/long line and finally 8 for Danish Seine.

There are significant differences (up till a factor 600) in the fuel consumption in the fishing stage depending on the target fish. Still, what appear to be even more interesting the fuel consumption varies considerably as a function of the fishing gear - even considering the same target species. In this article it is argued that a difference of a factor 15 exist between the fuel consumption per kg caught flatfish – depending on the type of fishing gear applied. It is argued that we need to address these differences in the search for a more fuel-efficient and sustainable fishery. In this regard, improvements in fuel efficiency appear to be consistent with other objectives, such as reduced impacts on sea floor habitats and reduced overexploitation of fish stocks

So far, the processing stage has been the focus of attention concerning the impacts on the external environment. They have been regulated through environmental approvals and a number of cleaner technology projects have been initiated through the 1990s. The focus has been water consumption and wastewater emissions. However, the LCA study suggests that the direct emissions of nutrients from the fish industry are insignificant due to wastewater treatment. This is a paradox and suggest that we need to draw more attention to the fishing stage – not only concerning overexploitation, but also in a wider environmental context that include considerations of how the fish are caught - not only how many.

References

Andersen, Erik et al. (1996): Overview of cleaner production practices in the fish industry, Miljøprojekt fra Miljøstyrelsen nr. 317, København K

Beam Trawler, 2001: Personal communication with the owner of three beam trawlers around 200 GT+ targeting European Plaice in the North Sea. Detailed catch composition for year 2000 have been available.

Christensen P. et al. (2001): LCA of Danish Fish products, Proceedings, International Conference on LCA in Foods, Gothenburg, Sweden 26-27 April, 2001, SIK-Dokument 143, Gothenburg

Danish Research Institute of Food Economics (2001): Account statistics for fishery 2000, Ministry of Food, Agriculture and Fisheries, Copenhagen.

ETH-ESU 96 (1996): Inventory data for Swiss and the Western European energy supply situation. Data include emissions from primary energy extraction and construction of infrastructures and energy conversion and transmission, The report and its database has been compiled, edited and revised by the ESU-group at ETH Zürich and the section of Comprehensive Systems Analysis at the PSI in Villigen.

European Environmental Agency (2001): EMEP/CORINAIR Emission Inventory Guidebook - 3rd edition. Technical report No 30. (Group 8: Other mobile sources).

Ekvall, T., and B. P. Weidema (2004): System boundaries and Input Data in Consequential Life Cycle Inventory Analaysis. International Journal of LCA. OnlineFirst.

Eyjólfsdóttir, H. R., E. Yngvadóttir, H. Jónsdóttir, and B. Skúladóttir. (2003): Environmental effects of fish on the onsumer´s dish - Life cycle assessment of Icelandic frozen cod products, Icelandic Fisheries Laboratories (IFL), Reykjavik, Island.

LCAfood (2003): LCA data on basic food products produced and consumed in Denmark, The present database has been created by Per H. Nielsen, Ph.D., Anne Merete Nielsen, M.Sc., Bo P. Weidema, Ph.D. from 2.-0 LCA consultants and Randi Dalgaard, M.Sc., Niels Halberg, Ph.D., Danish Institute of Agriculture Sciences, see www.lcafood.dk

Matcon A/S and Dansk Energi Analyse A/S, 1995: Branch energy analysis, Udarbejdet af Matcon A/S and Dansk Energi Analyse A/S for Energistyrelsen, Fiskeindustriens Arbejdsgiverforening og Dansk Industri, København.

Nielsen, R. (2002): Average fuel consumption and catch composition for different vessels, gear types and target species - year 2000, Danish Research Institute of Food Economics, Copenhagen.

Thrane, M. 2004b. Energy Consumption in the Danish Fishery - Identification of Key Factors. Journal of Industrial Ecology, 8(1): 223-239 <http://mitpress.mit.edu/catalog/item/default.asp?tid=12724&ttype=6>.

Thrane, M. (2004a): Environmental impacts forms Danish fish products, Ph. D dissertation, Department of Development and Planning, Aalborg University, Denmark. DRAFT

Thrane, M. (2004b): Energy Consumption in the Danish fishery – Identification of key factors Journal of Industrial Ecology.

Weidema, B. P. (2001): Avoiding co-product allocation in life-cycle assessment, Journal of Industrial Ecology 4(3):11-33.

Weidema, B. P. 2003: Market information in life cycle assessments. Environmental project no. 863. Danish Environmental Protection Agency, Copenhagen.

Wenzel, H. et al. (1997): Environmental Assessment of Products, Volume 1: Methodology, tools and case studies in product development, Chapman & Hall, London

Ziegler F et al. (2003): Life cycle assessment of frozen cod filets including fishery-specific environmental impacts, International Journal of life cycle assessment 8 (1) 39–47, ecomed publishers, Germany.

Operation specific engine load pattern and exhaust gas emission data from vehicles used in typical Swedish agricultural operations

M. Lindgren; O. Pettersson; O. Norén; P.-A. Hansson

** Department of Biometry and Engineering. P.O. Box 7032, SE 750 07 Uppsala, Sweden, tel: +46 18 67 12 39, fax: +46 18 67 35 29, magnus.lindgren@lt.slu.se;*
Swedish Institute of Agricultural and Environmental Engineering. P.O. Box 7033, SE 750 07 Uppsala, Sweden;

Abstract

The purpose of the work was to present operation specific engine load data, fuel consumption and exhaust gas emission data from vehicles used in typical agricultural operations. Three agricultural tractors and one combined harvester were equipped with instruments for measuring engine speed and loading torque. Recorded data were combined with emission data measured in a test bench in order to calculate operation specific fuel consumption and emission data. The result showed that the engine load patterns were rather independent of vehicle used but that the variations between different operations were big, both in terms of engine load pattern and resulting engine exhaust gas emissions.

Keywords: emission, fuel consumption, operation specific, agricultural

Introduction

High quality data on fuel consumption and engine exhaust gas emission amounts from agricultural tractors and other agricultural vehicles are needed in calculations of environmental loads caused from different food production and agricultural strategies (Hansson and Mattsson, 1999). Fuel consumption and engine exhaust gas emission data are normally produced through measurements according to different standards. In Europe, USA and Japan there is an obligation for manufactures to certify new engine models to emissions performance standards. The ISO 8178 (ISO, 1996) and ECE R49 (EEC, 2000) standards currently used for agricultural tractors are based on emission measurements under steady-state conditions. Emissions are measured individually at different modes (combination of engine speed and torque) and the resulting average emission values (one for each emission) are obtained by weighting the values of the different modes together. These standards are used for all types of nonroad mobile machinery (EU, 2000) and the weighting factors are not adapted to agricultural conditions (Treiber and Sauerteig, 1991: Rinaldi and Näf, 1992).

Due to the varying use of agricultural vehicles it is not possible to design one set of weighting factors valid for the average use of agricultural vehicles (Renius, 1994: Hansson *et al.,* 2001). Earlier studies (Hansson *et al.,* 2001) have shown that operational specific fuel consumption and emission amounts cannot be accurately calculated without account being taken to the actual engine load in the operation performed. Lindgren *et al.* (2003) found that the engine load strongly varied between different typical operations with agricultural tractors, from low load and engine speed operations like spreading fertiliser to high load and engine speed operations like a stubb puller operation. However, the amount of detailed operations specific fuel consumption and emission data available for these vehicles is very limited.

Most fuel consumption and emission data used today are obtained from measurements under steady-state conditions with no consideration to the engine‘s emission characteristics at transient conditions i.e. variations in engine speed and torque. Hansson *et al.* (2003) found that the fuel efficiency decreased with increasing proportion of fast variations in engine load while Lindgren *et al.* (2003) found that the emission amounts during a front end loading operation including fast load variations increased compared to steady-state conditions. Lindgren *et al.* (2003) also found that the variations in engine load for most operations with agricultural vehicles were rather slow. For most operations, except for high transient operations like front end loading, fuel consumption and emission amounts can be analysed with the semi-static approach as described by Hansson *et al.* (1998). This has also been confirmed by Cornetti *et al.* (1988).

The purpose of the work was to present operation specific fuel consumption and emission data from vehicles used in typical agricultural operations. The purpose was also to obtain operation specific engine load data for the vehicles mentioned.

Material and methods

Vehicles and instrumentation

Two medium-sized tractors, a Valtra 6600 and a Valtra 6650 with a rated engine power of 75 kW and 81 kW respectively, and one large tractor, a Case IH MX 270 with a rated engine power of 240 kW were studied. Furthermore, one combined harvester, a Massey Ferguson 7254 with a rated engine power of 160 kW, was included in the study. The Valtra 6600 tractor was manufactured in 1996, before engine exhaust gas emissions got regulated through emission performance standards. The Valtra 6650 and the Massey Ferguson were delivered in 2000 and 2002 and both complied with the European stage I regulation (EU, 2000). The Case IH MX 270 was delivered in 2002 and passed the stage II regulation. All vehicles were equipped with instrumentation for measuring engine speed, vehicle speed and fuel consumption. All data were sampled with 1 Hz and stored on a data logger. The measurement system is described in more detail in Pettersson *et al.* (2002).

Selection of operations

Six different farms, both with and without livestock handling, were studied. The main criterion was that the farm should have existing mechanical equipment suitable for the vehicle in question. The vehicles were delivered to the farm with the intention to replace the existing vehicles. Moreover, the drivers were instructed to use the vehicles in a normal manner. All the farms were situated in the middle of Sweden and the predominant textural class of soil was silty clay loams to clayey soils according to the Swedish soil materials e.g. more than 30 % clay (Hillel, 1982).

One individual operation for each type of operations was chosen as representative. The selection was based on statistical methods and calculated the fit of the individual operation to the sum of all operations of that specific operation. The operation with the best fit was chosen for further analysis and was considered to represent an as ‘normal’ operation as possible.

Emission measurements
The engines were tested at the Swedish Machinery Testing Institute located in Umeå, Sweden. All tests were conducted with a low sulphur (maximum 10-ppm sulphur) diesel fuel with low aromatic content, classified as Swedish environmental class 1 diesel fuel. Emissions of CO_2, CO, hydrocarbons (HC) and nitrogen oxides (NO_x) were measured as well as fuel consumption. Particulate matter emissions was only measured as an average value for the whole cycle and thus not possible to include in this study although particulate matter emissions could be substantial. The engine dynamometer used in the transient tests was a fast response Schenck eddy-current dynamometer with 400 kW maximum power. Engine output speed and torque was controlled through an electronic fuel pump rack control in combination with the dynamometer brake power. The engine dynamometer, control-system and analysis instruments are described in more detail in Wetterberg *et al.* (2002).

Fuel consumption and emissions of CO_2, CO, HC and NO_x were measured according to a 20 mode steady-state cycle. The 20 mode steady state cycle used was based on the 8 mode international ISO 8178 test cycle (ISO, 1996) and extended with 12 additional modes in order to increase the resolution, se Figure 1. The fuel consumption and emissions were measured in accordance with the ISO 8178 regulation (ISO, 1996).

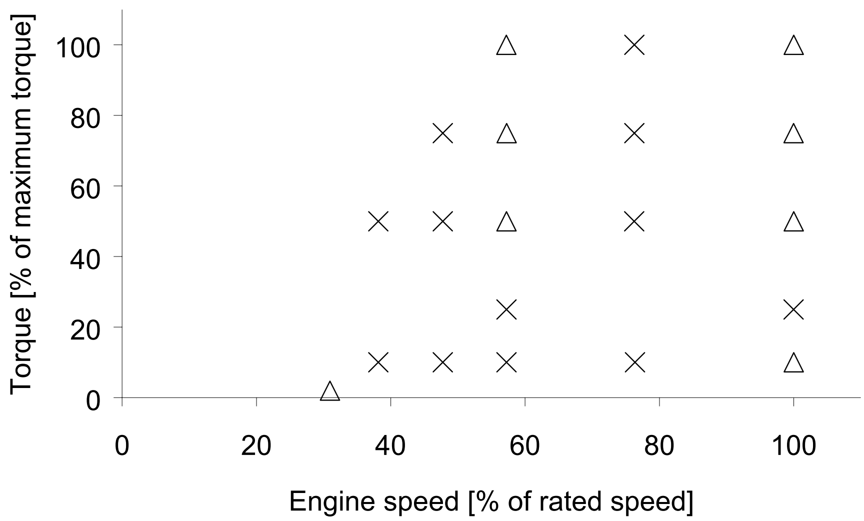


Figure 1. 20-mode steady-state test cycle based on ISO 8178 (Δ) and extended with 12 additional modes (X).

Data analysis

The data analysis was based on the following parts:

(1) The development of an expression deciding the engine torque in Nm (moment) as a function of engine speed and fuel consumption. The proposed model was similar to that expressed by Jahns (1983) and consisted of nine coefficients specific to the engine:

$$\tau = c_1 \times f + c_2 \times f^2 + c_3 \times f^3 + \left(c_4 \times f + c_5 \times f^2 + c_6 \times f^3\right) \times n + \left(c_7 \times f + c_8 \times f^2 + c_9 \times f^3\right) \times n^2$$

where τ was engine torque, n was engine speed, f was steady state fuel consumption and c_i were engine specific coefficients. The coefficients c_1 to c_9 were found for each engine by least-squares regression (R^2=1.00) using values of engine speed, engine torque and steady-state fuel consumption obtained from the emission measurements;

(2) The development of a matrix over emissions of CO_2, CO, HC and NO_x for all possible steady-state combinations of engine speed and torque data for each engine, within the operational range of the engine. Engine speed and torque data were rounded towards the nearest integer. The data for the matrix were obtained from the 20-mode steady-state emission measurement by two-dimensional interpolation;

(3) Registration of engine speed, fuel consumption etc when performing the work operation to be analysed;

(4) Calculation of the engine torque using the recorded engine speed and fuel consumption data from part (3) as inputs to the expression from part (1);

(5) Calculation of operation specific weighting factors by dividing the operational range of the engine into eight smaller areas, one for each mode in the ISO 8178 standard (Hansson et al., 1999). The weighting factors were calculated as the relative frequency of recorded combinations of engine speed and torque data in each of the eight areas.

(6) Calculation of operation specific emission data using the time series of engine speed and torque data from part (4) as inputs to the matrix in part (2);

Result

Operation specific engine load pattern

Engine load patterns were recorded for several different operations and varying conditions. In Table 1 and 2 is the engine load pattern presented as weighting factors according to the ISO 8178 standard for recorded operations with the agricultural tractors and the combined harvester, respectively. The variations in engine load between different operations was substantial *e.g.* from fairly steady-state, low engine speed and low load operations like urine manure spreading and the PTO driven tedder operation to operations with highly varying engine load like the transport operations or the power demanding precision chopper operation.

Operation specific emissions

The operation specific fuel consumption and emission data for the operations recorded are presented in Table 3 and 4. Data of average engine power and cultivated area per hour, where required, is also shown in Table 3 and 4. Table 3 includes multiples of several operations depending on the different vehicles used. Each vehicle, e.g. engine, produces different amounts of emissions due to different technical designs etc. Examples of the variation in fuel consumption and emission amounts between different operations are presented in Figure 2. The data are presented as relative values compared to the ISO 8178 amounts.

The result presented in table 3 and 4 can easily be recalculated to g l^{-1} or g MJ^{-1} fuel consumed by use of the density (814 g l^{-1}) and energy content of diesel (35.15 MJ l^{-1}).

Table 1. Operation specific engine load data for typical operations with agricultural tractors.

Operation	Distribution of engine operations by ISO 8178 modes in %							
	Mode 1	Mode 2	Mode 3	Mode 4	Mode 5	Mode 6	Mode 7	Mode 8
ISO 8178 C1	15	15	15	10	10	10	10	15
Bale wrappers	0	0	0	0	1	7	40	52
Baler	0	32	10	0	0	1	56	1
Fertiliser spreader	1	6	4	1	1	1	43	43
Forest trailer	4	4	2	0	2	2	6	80
Front end loading	0	0	2	8	0	0	73	17
Harrowing, heavy	56	1	0	0	23	16	2	2
Harrowing, normal	27	47	7	0	3	11	5	0
Mower conditioner, heavy	38	24	2	0	1	35	0	0
Mower conditioner, light	0	0	0	1	0	1	95	3
Ploughing	61	8	6	9	6	2	5	3
Ploughing	38	13	8	6	8	9	18	0
Precision chopper, heavy	89	10	1	0	0	0	0	0
Rolling	0	0	1	0	0	2	83	14
Semi-liquid manure spreader	0	36	63	1	0	0	0	0
Solid manure spreader	13	26	26	1	0	18	16	0
Sowing, high engine speed	1	25	67	4	0	0	3	0
Sowing, low engine speed	0	3	3	0	0	7	79	8
Stubb puller	73	26	1	0	0	0	0	0
Tedder	0	0	0	0	0	1	99	0
Transport of gravel	13	7	12	5	4	6	12	41
Transport of manure	44	13	17	4	16	1	4	1
Transport on country road	37	27	14	8	7	3	2	2
Urine manure filling	0	0	10	40	0	0	40	2
Urine manure spreader	0	1	2	1	0	0	95	1

Table 2. Operation specific engine load data for typical operations with a combined harvester.

Operation	Distribution of engine operations by ISO 8178 modes in %							
	Mode 1	Mode 2	Mode 3	Mode 4	Mode 5	Mode 6	Mode 7	Mode 8
ISO 8178 C1	15	15	15	10	10	10	10	15
Barley harvesting	6	26	50	18	0	0	0	0
Oats harvesting	4	23	49	24	0	0	0	0
Rapeseed harvesting	5	22	54	19	0	0	0	0
Winter wheat harvesting	18	44	28	10	0	0	0	0

Table 3. Operation specific fuel consumption and emission data for typical operations with agricultural tractors.

Operation	Power	Work rate	Fuel	Emissions (g/h)			Vehicle
	%	ha/h	kg/h	CO	NO_x	HC	
Bale wrappers	27		5.0	22.3	236	5.9	Valtra 6600
Baler	56		9.5	22.9	369	9.3	Valtra 6600
Fertiliser spreader	21	11.0	4.4	32.3	181	7.2	Valtra 6600
Forest trailer	2		3.0	36.1	148	6.2	Valtra 6600
Front end loading	16		3.8	22.6	77	6.6	Valtra 6650
Harrowing, heavy	90	4.9	14.8	99.8	716	10.3	Valtra 6650
Harrowing, normal	76	8.0	12.8	27.0	469	10.2	Valtra 6600
Mower conditioner, heavy	79	2.5	12.9	25.8	512	9.4	Valtra 6600
Mower conditioner, light	33	1.6	6.8	24.3	172	9.1	Valtra 6650
Ploughing	68	0.7	11.6	27.5	449	9.4	Valtra 6600
Ploughing	80	0.6	13.5	88.9	624	10.2	Valtra 6650
Ploughing	70	1.9	34.4	107.8	1202	11.7	Case IH
Precision chopper, heavy	99	0.8	17.3	30.8	618	10.6	Valtra 6600
Rolling	37	5.8	6.9	17.2	216	7.5	Valtra 6650
Semi-liquid manure spreader	53	6.9	10.3	29.5	310	11.4	Valtra 6650
Solid manure spreader	59	2.4	10.5	33.6	385	10.4	Valtra 6650
Sowing	48	6.6	24.3	79.0	679	11.2	Case IH
Sowing, high engine speed	52	2.3	9.5	29.0	311	10.7	Valtra 6600
Sowing, low engine speed	41	2	7.6	21.2	243	8.4	Valtra 6650
Stubb puller	92	1.3	15.9	30.0	564	10.7	Valtra 6600
Tedder	32	2.2	6.2	25.8	241	7.9	Valtra 6600
Transport of gravel	35		6.4	33.3	257	7.9	Valtra 6600
Transport of manure	79		13.3	87.5	596	10.8	Valtra 6650
Transport on country road	72		12.3	62.5	496	10.9	Valtra 6650
Urine manure filling	15		4.1	40.4	140	8.8	Valtra 6600
Urine manure filling	22		5.9	28.0	102	9.5	Valtra 6650
Urine manure spreader	37	4.4	7.0	25.0	267	8.6	Valtra 6600
Urine manure spreader	35	6.7	6.7	23.5	181	8.5	Valtra 6650

Table 4. Operation specific fuel consumption and emission data for typical operations with a combined harvester.

Operation	Power	Work rate	Fuel	Emissions (g/h)			
	%	ha/h	kg/h	CO	NO_x	HC	Vehicle
Barley harvesting	48	1.6	22.9	211	739	12.6	MF 7254
Oats harvesting	44	1.8	21.7	191	675	12.1	MF 7254
Rapeseed harvesting	44	2.4	21.2	175	680	11.5	MF 7254
Winter wheat harvesting	63	2.1	27.8	337	1045	14.7	MF 7254

Discussion

Standardised test cycles like ISO 8178 shows a fairly uniform engine load with slightly emphasis on the high speed and load area. Only a minority of the typical agricultural operations studied had an engine load pattern that resembled the standardised one, *i.e.* on-road transport with universal trailer. The results clearly shows that the standardised weighting factors and thus the resulting fuel consumption and emission data is not adapted to the real use of the tractor. Therefore, the use of one single set of engine load data, *i.e.* weighting factors, and fuel consumption and emission data valid for all operations with typical agricultural vehicles, as for example in life cycle assessment analyses (LCA), may give misleading results.

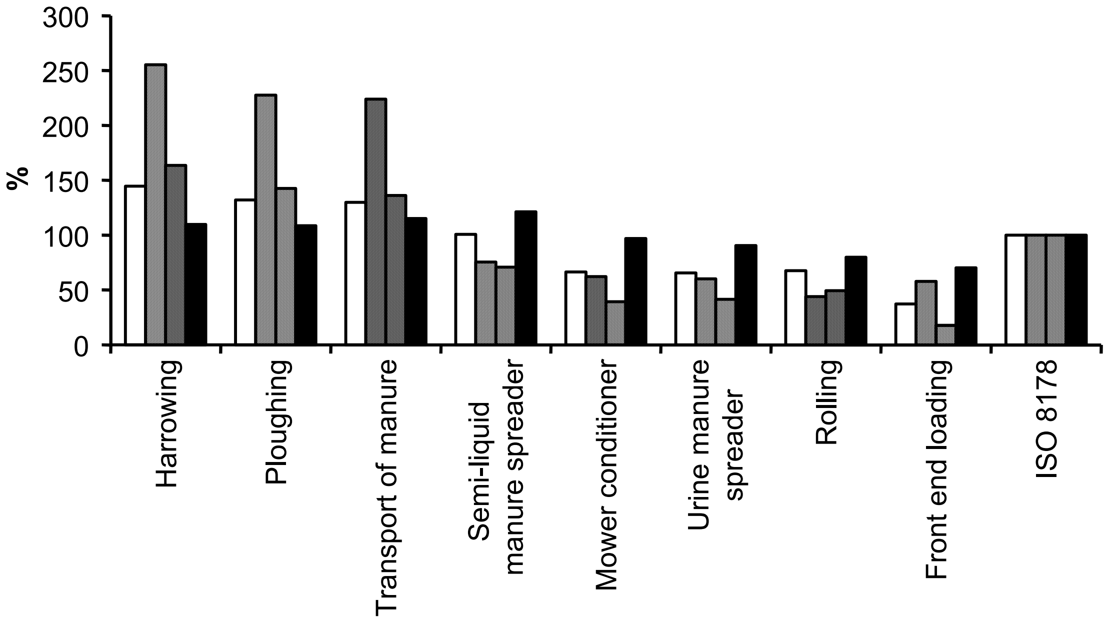


Figure 2. Fuel consumption and emission data for different operations with the Valtra 6650 agricultural tractor relative to the ISO 8178 standard measurement: □, fuel consumption; ■, CO; ■, NO_x; ■, HC.

The standardised measurement method used today for nonroad mobile machinery, ISO 8178, is based on measurements at steady-state load conditions. When measure at steady-state conditions no respect is take to the engine's characteristics at fast variations in engine speed and torque. Resent research work has shown that transients in engine speed and torque strongly influence the fuel consumption and emission amounts (Lindgren *et al.* 2003). However, most operations studied in this work were of less transient nature. For operations frequently including fast variations in engine speed and load, such as front end loading, the effects of transients are considerable and have to be included. Furthermore, a transient test cycle, 'nonroad transient cycle', is under consideration for use in forthcoming emission regulations, stage III B in Europe (EU, 2002) and tier 4 in USA (US EPA, 2003). A transient test cycle includes sections of acceleration as well as steady-state conditions and emissions are measured continuously over the whole transient test cycle.

Emission values published for different diesel engines show rather wide variations, especially when comparing a non-regulated stage 0 engine with a new engine that comply with the stage 2 emission regulation. However, the engine load patterns presented within this study showed that the engine load pattern were rather independent of vehicle used, a harrowing operation with the Valtra 6650 and the Case IH MX tractor were almost the same in terms of engine load pattern. The engine load for a specific operation varied slightly depending on the size of the implement compared to the engine power available.

The engine load patterns presented in this study can be used to calculate operations specific fuel consumption and emission data also for vehicles with engines and fuels other than these in the original measurements. In order to obtain operations specific fuel consumption and emission data for a new vehicle just multiply the individual mode of the ISO 8178 test for the new vehicle with the corresponding weighting factors for the operation in question.

Table 5 shows an example of how the fuel consumption data measured according to the standardised ISO 8178 regulation can be adjusted to better represent the operation in question, harrowing. Operation specific emission data for other vehicles or fuels than these in the original measurements can be obtained with the same technique.

Table 5. Calculation of operation specific fuel consumption and emission data from an ISO 8178 test and operation specific engine load data.

Mode	Fuel consumption kg/h	Weighting factor (%)		Fuel consumption (kg/h)	
		ISO 8178	Harrowing	ISO 8178	Harrowing
1	17.7	15	56	2.7	9.9
2	13.6	15	1	2.0	0.1
3	10.0	15	0	1.5	0
4	4.6	10	0	0.5	0
5	14.1	10	23	1.4	3.2
6	10.5	10	16	1.1	1.7
7	7.4	10	2	0.7	0.2
8	0.8	15	2	0.1	0
Sum		100	100	10.0	15.1

References

Cornetti, G., Klein, K., Fränkle, G. J. and Stein, H. J., 1988. US transient cycle versus ECE R49 13-mode cycle. *Society of Automotive Engineers (SAE)*, Paper 880715.

EEC, 2000. Uniform provisions concerning the approval of compression-ignition (C.I.) and natural gas (NG) engines as well as positive-ignition (P.I.) engines fuelled with liquefied petroleum gas (LPG) and vehicles equipped with C.I. and NG engines and P.I. engines fuelled with LPG, with regard to the emissions of pollutants by the engine. Regulation NO. 49, Revision 3, E/ECE/324, The European Union.

EU, 2000. Directive 2000/25/EC of the European Parliament and of the Council of 22 May 2000 on action to be taken against the emission of gaseous and particulate pollutants by engines intended to power agricultural or forestry tractors and amending Council Directive 74/150/EEC. *European Union Official Journal* L 173, pp 1-34.

EU, 2002. COM(2002) 765 final 27.12.2002, Proposal for a Directive of the European Parliament and of the Council amending Directive 97/68/EC on the approximation of the Laws of the Member States relating to measures against the Emission of Gaseous and Particulate Pollutants from Internal Combustion Engines to be Installed in Non-Road Mobile Machinery. The European Union.

Hansson, P-A., Norén, O. and Bohm, M., 1999. Effects of specific operational weighting factors on standardized measurements of tractor engine performance. *Journal of Agricultural Engineering Research* **74**, 347-353.

Hansson, P-A., Burström, A., Norén, O. and Bohm, M., 1998. Bestämning av motoremissioner från arbetsmaskiner inom jord och skogsbruk [Engine emissions from agricultural tractors and forestry machines]. Department of Agricultural Engineering, Uppsala, Sweden, Report 232.

Hansson, P-A., Lindgren, M., Nordin, M. and Pettersson, O., 2003. A methodology for measuring the effects of transient loads on the fuel efficiency of agricultural tractors. *Applied Engineering in Agriculture,* **19**(3), 94-100.

Hansson, P-A., Lindgren, M. and Norén, O., 2001 A Comparison between different methods of calculating average engine emissions for agricultural tractors. *Journal of Agricultural Engineering Research*, **80**(1), 37-43, doi: 10.1006/jaer.2001.0710.

Hansson, P-A., and Mattsson, B., 1999. Influence of derived operation-specific tractor emission data on results from an LCI on wheat production. *The International Journal of Life Cycle Assessment*, 1999. **4**(4), 202-206.

Hillel, D., 1982. Introduction to soil physics. Academic press limited, London ,UK.

ISO, 1996. ISO 8178-4; Reciprocating internal combustion engines- Exhaust emission measurements- Part 4: Test cycles for different engine applications. International Organisation of Standardisation.

Jahns, G., 1983. A method of describing diesel engine performance maps. *ASAE* Paper No 83-103, St Joseph, Mich.: ASAE.

Lindgren, M., Pettersson, O., Hansson, P. and Norén, O., 2003. Jordbruks- och anläggningsmaskiners motorbelastning och avgasemissioner samt metoder att minska bränsleförbrukning och avgasemissioner [Engine load pattern and engine exhaust gas emissions from off-road vehicles and methods to reduce fuel consumption and engine exhaust gas emissions]. Report Agriculture and Industry 308, Swedish Institute of Agricultural and Environmental Engineering.

Pettersson, O., Norén, O., Hansson, P-A. and Lindgren, M., 2002. A system for onboard determination of engine powerby measuring fuel consumption at 1 Hz. EurAgEng Budape2002. Paper Number 02-PM-024.

Renius, K. T., 1994. Trends in Tractor Design with Particular Reference to Europe. *Journal of Agricultural Engineering Research*, **57**(1), 3-22.

Rinaldi, M. and Näf, E., 1992. Motorauslastung von Landwirtschaftstraktoren (Engine loads on agricultural tractors. *FAT-berichte* No. 426. Tänikon, Switzerland.

Treiber, P. and Sauerteig, J., 1991. Present and future European exhaust emission regulations for off-road diesel engines. *Society of Automotive Engineers (SAE),* Paper 911808.

US EPA, 2003. Notice of proposed Rulemaking. Code of Federal Regulation 40 Part 69, 80, 89, 1039, 1065, and 1068. Control of Emissions of Air Pollution from Nonroad Diesel Engines and Fuel. USA Environmental Protection Agency.

Wetterberg, C., Holmgren, K., Rosell, L., Johansson, L. and Magnusson, R., 2002. The influence of the fuel on emissions from diesel engines in large off-road machines. Report AL90B 2002:16295, Swedish National Road Administration, Borlänge, Sweden.

An LC inventory based on representative and coherent farm types

Randi Dalgaard[1], Niels Halberg[1], Ib Sillebak Kristensen[1] and Inger Larsen[2]

[1]*Danish Institute of Agricultural Sciences, Department of Agroecology, Research Centre Foulum, P.O. Box 50, DK-8830 Tjele. Tel: +45 8999 1254 - Fax: +45 8999 1200. E-mail: Randi.Dalgaard@agrsci.dk.* [2]Danish Research Institute of Food Economics

Abstract

There is a need for valid and representative data regarding the production, resource use and emissions from typical farming systems in Denmark for analysis of the environmental impact of different systems and as input to product oriented analyses such as Life Cycle Assessments of basic food items. An inventory of 31 farm types was constructed on the basis of 2138 farm accounts from 1999 selected and weighted to be representative for the Danish farming sector. The farm accounts were grouped according to the major soil types, the number of standard working hours, the most important enterprise (dairy, pig, different cash crops) and the stocking rate (livestock units per hectare). For each group the account data on the average inputs and outputs, land use and herd structure was used to establish a farm type model with coherency between livestock production, total feed use, land use, yields, imported feed, home-grown feed, manure production, fertiliser use and crop production. The set of farm types were scaled up to national level thus representing the whole Danish agricultural sector for the included products. The sum of area and yield by crop, number and production by livestock type and the use of fertiliser, energy and concentrated feed was checked against national level statistics and corrected accordingly across all farm types. Resource use and emissions in each farm type was established using standard nutrient concentrations and models for nutrient cycling, energy use and emissions of e.g. ammonia, nitrous oxides and methane. For LCA the product oriented inventory was established using system expansion rather than allocations to account for the secondary enterprises in the livestock farm types. Data are made available on a web-based database and may be used for analyses of the primary production systems or as input for LCA across the whole production chain.

Background

For most products the primary agricultural food production is an important determinant of the total resource use and environmental impact, which is why life cycle assessment (LCA) of food products must carefully address the question of data quality for agricultural production.

Many existing Inventories for LCA of agricultural products have used case studies, based on recordings on a limited number of farms. However, there is a large variation in the resource use and environmental impact between farms with the same main enterprise (Halberg, 1999; Weidema et al., 2002). Thus, an LCA that aims at a more general validity must be based on a larger sample of farm data being representative for the systems in question (average or marginal depending on the purpose of the LCA) and preferably be checked against statistical information from the level the sample will represent (e.g. regional or national).

This paper present an LCI which is based on representative farm accounts and is used to model the input and production of typical farms using a method that allows to check that the models are consistent with higher level statistical information following ideas described by Halberg et al. (2000).

Objective

The objectives of this paper is:

- To present a method for establishing LCI for important farm types based on representative data for the Danish agricultural sector and farm models.
- To give examples of LCI data and discuss problems and advantages in using representative statistical farm data for LCI.

Methods

All Danish farms are obliged to keep detailed records of purchases and sales for tax purposes and the yearly accounts are made with professional help. A representative data set of these accounts, 2138, are reported by the advisors to the Danish Research Institute of Food Economics (DRIFE) and constitute the basic empirical data input to the model of representative farm types presented here. The accounts include besides economical data, technical data on the land use, livestock numbers in different groups and cash crop yields including cereals. The representative data set was based on farm accounts from 1999, sampled as to present the total Danish agricultural sector of the main livestock and crop production. Thus, each farm account is given a weight to allow for division into sub-populations/groups and for scaling up the sample to national level (Larsen, 2003).

The accounts in the data set were divided into 31 groups. Each group contained from 5 to 185 accounts and represented one of the 31 farm types according to soil type (loamy vs. sandy), main enterprise (dairy, beef, pig, poultry and different cash crop types), organic vs. conventional and animal density (e.g. livestock units per ha). For each farm type a detailed model was established partly based directly on the average accounts data within each group and partly on general knowledge as explained in the following: Step 1: Modelling coherent farm types which have a realistic balance between crop and livestock production, use of inputs and sale of products. Step 2: Modelling the emissions (CH_4, CO_2, NO_3, HPO_3, NH_3 and N_2O) from the individual farm types.

Step 1. Modelling farm types

The average partition of land with different crops and the number of livestock in each group was used to establish the production of each farm type. The accounts also gave information on crop yields and amounts of cash crops (e.g. cereals, rape seed, potatoes, grass seeds) and livestock products sold (milk, meat, live animals). This information was thus used to establish the level of production within each type and the general crop-livestock interaction (e.g. how much grassland was used for cattle). However, because the use of external inputs like purchased feed and fertiliser was only available in monetary units the exact feed and fertiliser use was modelled using standards. Due to the public regulation of manure and fertiliser use in Denmark representative average values for feed efficiency in livestock production (e.g. feed use per kg live weight pig) and the production of Nitrogen (N) and Phosphorus (P) in manure by livestock types is well established (Poulsen et al., 2001). Moreover, each farm has a fertiliser quota based on official crop-N norms deducted the plant available manure-N produced or imported. Thus, the fertiliser use on the farm types was calculated using these norms. As part of Danish compliance with the Nitrates Directive the use of manure-N is limited (e.g. 140 kg manure N per ha on pig farms) why some farms are obliged to export manure to cash crop farms. This was modelled as transfer of manure from farm types with high stocking rate to other types, which then reduced the fertiliser input accordingly.

This way a coherent model of crop-livestock interactions was established for each farm type with a consistent relation between livestock production, use of home-grown vs. imported feed and export of cash crops. Energy use for traction was modelled following Dalgaard et al. (2001) where each crop is assigned a number of field operations multiplied by diesel use per ha. Electricity use was estimated directly from the accounts. The total land use and yields of each crop, the number of livestock, imported feed and fertiliser etc. across all farm types were then checked against national level statistical information (Agricultural Statistics, 2000) to make sure that the typology as whole was consistent and representative for Danish agricultural sector. As shown in table 1 the data set based on farm accounts is in good agreement with the Danish national statistics (Agricultural Statistics, 2000) for land use and for pig and milk production. The total area and yield of major cash crops (not shown) also fits to national statistics.

Table 1. Selected data from the typology of farm models scaled up to national level and compared with the Danish national statistics (Agricultural Statistics, 2000)

	Typology of farm types	Danish national statistics	Deviation from nat. stat.
Slaughtering pigs [1] produced, 1000	20639	20801	-1%
Sows (yearly basis) 1000	1083	1052	3%
Milking cows[2], 1000	633	661	-4%
Milk production, 1000 tons	4624	4455	4%
Agricultural area, 1000 ha	2585	2644	-2%
Area with cereals, 1000 ha	1395	1448	-4%
Area with roughage, 1000 ha	567	570	-1%
Fertiliser N, 1000 tons N	226	252	-10%
Soybean meal, 1000 tons N	142	156	-9%
Grain, 1000 tons	6571	6728	-6%
Diesel and fuel, PJ	13	14	-18%

[1] Living weight = 100 kg

[2] Milking cows

The typology of farm models did, however, not account satisfactorily for the total use of fertiliser. Therefore, the farm models were adjusted using some of the slack in the determination of individual fertiliser quotas per farm and finally the still unexplained difference was corrected using an overall factor on the input to all farm types. The model also underestimated the total use of diesel and fuel by 20%, and therefore the farm models were adjusted accordingly.

Use of medicine is not considered and pesticide use was not included in the first version. Resource use and emissions related to the construction and maintenance of buildings and machinery used on the farm was not included.

Step 2. Modelling emissions
The emissions of gasses and other substances relevant for LCA impact categories were calculated based on the established resource use and production including land use and herd structure. The emissions of green house gasses were calculated using standard IPCC methodology for methane production from livestock and nitrous oxide production from soils and all relevant manure and fertiliser compartments (IPCC, 1997; 2000). Following the TIER 2 of the IPCC principles specific data for Danish crops and manure handling were used. The CO_2 emission was calculated from the use of fossil fuel for traction and stables. Emissions related to the production of farm inputs like fertiliser and soybean meal, which happen outside the farm may be included in a second step and have been established as separate processes in the LCI database (Nielsen et al., 2003).

Emissions of nitrate for the eutrophication/nutrient enrichment impact category was assumed to be equal to the farm gate balance minus ammonia losses, denitrification (Kristensen et al., 2003) and net change in soil N status. The ammonia emission from stables, manure storage and handling was calculated using standard values from Andersen et al. (1999). Denitrification was estimated using the method of Winter (2003), and net change in soil N status was modelled using the method of Petersen et al. (2002).

Table 2 shows the aggregated emissions over all farm types compared with national statistics for emissions of green house gasses (Gyldenkærne et al., 2004) and ammonia (Andersen et al., 2001). The difference in nitrous oxide emission was expected since we used more detailed information regarding crop residues than in the national nitrous oxide budget. The methane emission was 10% lower and the ammonia emission was 1% lower than national statistics.

Table 2. Selected emissions from the typology of farm models scaled up to national level and compared with the Danish national statistics (Andersen et al., 2001; Gyldenkærne et al., 2004,)

	Typology of farm types	Danish national statistics	Deviation from nat. stat.
N_2O (1000 tons)	22	20	9%
CH_4 (1000 tons)	160	177	-10%
NH_3 (1000 tons N)	76	77	-1%

Results

The resulting 31 farm type models after correction for national level consistency shows inputs and outputs used to produce specific amounts of livestock and cash crop products with different land use according to major enterprise and livestock density. Detailed results are presented at an open database (Nielsen et al., 2003). Table 3 shows a part of the inputs and outputs associated with production at the different dairy farm types.

Table 3. Main characteristic, inputs and outputs associated with agricultural production at eight different dairy farm types. Data are provided per farm per year.

Farm type	4	5	6	7	16	17	18	19
Characteristics								
Soil type	Loamy (clay)				Sandy			
Stocking rate (Livestock Units/ha)	<1.4	1.4-2.3	>2.3	Organic farms	<1.4	1.4-2.3	>2.3	Organic farms
Pct. of Danish farms[3]	0.9	1.7	5.3	0.2	3.8	7.9	0.7	1.4
Number cows	55	55	82	62	48	67	76	85
Land area (ha)	99	50	44	88	81	65	48	102
Milk yield per cow per year	7227	7288	7053	6811	7431	7429	7125	6866
Pct. of total Danish milk production	4	7	3	1	15	43	4	9
Pct. of cows' feed produced on farm	83	64	36	74	85	66	42	71
Inputs								
Soybean meal, tons	59	70	168	15	49	77	125	24
Spring barley, tons	0	65	177	104	0	92	211	154
Fertiliser, kg N	10689	4486	2096	0	8806	6602	3580	0
Fertiliser, kg P	1016	554	0	0	872	909	758	0
Diesel, MJ	515111	292549	326952	384807	409783	376043	336181	439502
Electricity, kWh	46190	30003	44258	39399	34929	42162	45563	55127
Outputs								
Milk, tons	399	398	576	424	355	499	538	583
Bread wheat, tons	76	17	34	27	37	12	8	8
Beef meat, tons	25	15	20	16	20	21	24	18
Rape seed, tons	8	1	0	0	6	0	0	0

More than 50% of the total Danish milk was produced on the sandy soil types with low and medium stocking rate. There are differences in farm size and the percentage of feed produced on farm between the types. Farm types with high stocking rate produce a smaller part of the feed on the farm and import more soybean meal compared to farm types with lower stocking

[3] Percentage of Danish farms represented by the farm type

rate. The average organic farm is larger than the conventional farm types, has lower milk yield per cow and crop yields per ha and produces more feed on the farm, especially based on grass-clover leys in crop rotation with cereals. In the model the organic farm import around 20 kg N per ha in manure from conventional farms.

The resulting environmental impact per kg milk produced at farm-gate after system expansion and displacement of cash crops is shown in table 4. Milk produced at farm types with low stocking rates (farm type 4 and 16) shows a tendency to lower environmental impact than milk produced at farm types with medium stocking rates (farm type 7 and 17). The farms with high stocking rate have to export manure according to public regulation, which decreases emissions from the farm. Land use per kg milk increase with higher stocking rate, because the land used for imported feeds are involved.

Similar results for pig meat and major cash crops on farm level and per kg product ex farm are presented by Nielsen et al. (2003) on the open database: www.lcafoods.dk.

Table 4. Environmental impact from production of 1 kg of milk from six different conventional dairy farm types

	Units (eqv.)	Farm type 4	Farm type 5	Farm type 6	Farm type 16	Farm type 17	Farm type 18
Global warming	g CO_2	754	910	726	943	1030	998
Eutrophication	g NO_3	14.3	36.2	22.7	46.9	52.3	50.6
Acidification	g SO_2	7.6	9.6	10.1	9.0	10.0	10.9
Photochemical smog	g ethene	0.25	0.26	0.24	0.27	0.28	0.30
Land use	m^2year	1.18	1.36	1.48	1.31	1.38	1.57

Discussion and conclusion

The present LCI is based on realistic levels of resources used per unit of produced product and reflects average production levels and efficiency within different farm types. The types are all consistent in terms of crop-livestock interactions. The typology accounts for most input and output of the Danish agricultural production including the exchange of manure between farm types. The factors soil types and livestock density were assumed to be the primary systematic determinants of the level of resource use and emissions from farms. The farm models allow for the calculation of emissions per kg of product using system expansion and displacement as demonstrated by Nielsen et al. (2003). The process of system expansion is, however, not straightforward and involves critical assumptions regarding marginal producers of the avoided products. In the case of exported manure the methodological choice of using system expansion gave a different allocation of ammonia losses than is often used when comparing nutrient balances and losses from farming systems (e.g. Kristensen et al., 2003). Because the exported manure only displaces an amount of N fertiliser equal to the plant available N content (i.e. the part of total manure-N taken up by the crop when compared to fertilizer in trials) the ammonia losses from spreading the manure on the importing farm is still included in the emissions of the manure producing farm.

The basis for the established typology of farm models is a set of representative farm accounts on the form that is used for statistical purpose including the Danish reporting to the Farm Account Data Network (FADN), which again forms part of EU agricultural statistics (Poppe et al., 2000). Thus, this type of data will be available for most European countries, which again could facilitate the development of more uniform methods for LCI establishment across different countries. Another advantage of this method is that it may be updated relatively easily with data for the subsequent years when accounts data are available.

The major drawback of the method from the authors' point of view is that the large variation between farms in e.g. feed or fertiliser use efficiency due to differences in farmers' management skills and strategic choices regarding crop rotation and feed planning is not reflected in differences between the farm types. This was, however, a necessary choice based on the primary purpose: To get representative and statistically valid data for an LCI to be used for comparison of different products and securing a valid baseline for LCA on processed food products. The amounts of feed and fertiliser purchased could have been modelled based on the monetary information using standard prices per unit but that might have introduced another bias because of differences in the actual price per unit paid (e.g. large farms that get discount prices would in reality have used more feed or fertiliser than estimated from average prices). One hypothesis could be that farmers in the marginal types would be more efficient than the average farmers and thus have a lower resource use and emissions per kg product delivered. The results show differences in resource use and emissions per kg product between farm types, but more sensitivity analyses are needed in order to determine if these differences are significant.

Another drawback is the relatively large number of small co-enterprises in the farm types resulting from combining a large number of farm accounts with different co-enterprises (e.g. two dairy farms growing five hectares with cash crops, one bread wheat, the other sugar beets will result in a type growing 2.5 hectares of each). This results in a number of co-enterprises that have to be compensated for through system expansion. A solution to this would be to eliminate some of these co-enterprises in the modelled farm types, which however further would detach the model from the empirical data.

The typology did not initially account for the total use of fertiliser in Danish agriculture why a correction factor was used. While this secures consistency with national level statistics it is not a totally satisfactory solution because the error may in fact belong to underestimation in specific rather than all types. Fertiliser use in Danish farms is strongly regulated presently and it was considered most realistic to adjust all farm types equally in order to fit the national statistic.

It can be concluded that the resulting LCI demonstrates successfully a method to establish coherent and representative inventories of agricultural production based on generally available data.

Reference

Agricultural Statistics. 2000. Statistik om landbrug, gartneri og skovbrug. Danmarks Statistik. ISBN 87-501-1195-7. 287 pp.

Andersen, J.M., Sommer, S.G., Hutchings, N.J., Kristensen, V.F. and Poulsen, H.D., 1999. Emission af ammoniak fra landbruget - status og kilder. Ammoniakfordampning - redegørelse nr. 1, Danmarks JordbrugsForskning, 63 pp.

Andersen, J.M., Poulsen, H.D., Børsting, C.F., Rom, H.B., Sommer, S.G. and Hutchings, N.J. 2001. Ammoniakemission fra landbruget siden midten af 80'erne. Fagligt rapport fra DMU nr. 353. 47 pp.

Dalgaard, T., Halberg, N. and Porter, J.R., 2001. A model for fossil energy use in Danish agriculture used to compare organic and conventional farming. *Agriculture, Ecosystems & Environment* **87(1):** 51-65.

Gyldenkærne, S. and Mikkelsen, M.H. 2004. Projection of Greenhouse Gas Emission from the Agricultural sector. Research Notes from NERI No. 194, National Environmental Research Institute. Denmark. 50 pp.

Halberg, N., 1999. Indicators of resource use and environmental impact for use in a decision aid for Danish livestock farmers. *Agriculture, Ecosystems & Environment* **76** 17-30.

Halberg, N., Kristensen, I.S. and Dalgaard, T., 2000. Linking data sources and models at the levels of processes, farmtypes and regions. Agricultural data for Life Cycle Assessments. Series 2.00.01, vol.1. Agricultural economics research Institute (LEI), The Hague, the Netherlands. p. 16-30. ISBN 90-5242-563-9.

IPCC, 1997. Greenhouse gas inventories. Revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories. Available on-line: http://www.ipcc.ch/pub/guide.htm

IPCC, 2000. IPCC Good practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories. Chapter 4. Agriculture. 4.1-4.83. IPCC. Available on-line: http://www.ipcc.ch/pub/guide.htm

Kristensen, I. S., Halberg, N., Nielsen, A. H., Dalgaard, R., and Hutchings, N., 2003. N-turnover on danish mixed dairy farms, 1-21. Paper for workshop: *"Nutrient management on farm scale: how to attain European and national policy objectives in regions with intensive dairy farming?"*. 23-25 June 2003, Quimper, France. Available on-line: http://www.agrsci.dk/jbs/isk/DK_country_report_partII.pdf.

Larsen, I., 2003. LCA project. Method description. Available online, 1/10 2003 at www.lcafood.dk/database.

Nielsen, P.H., Nielsen, A.M., Weidema , B.P., Dalgaard, R. and Halberg N., 2003. LCA food database. Available online, 31/8 2003 at www.lcafood.dk/database.

Petersen, B.M. & Berntsen, J., 2002. Omsætning i jordpuljen på forskellige bedriftstyper. Temadag arrangeret af Afd. for Jordbrugssystemer 24. april 2002. Forskningscenter Foulum. Intern rapport nr. 157: 13-24.

Poppe, J.K. and Meeusen, J.G., 2000. Using a Farm Accountancy Data Network in data management for LCA. Agricultural data for Life Cycle Assessments. Series 2.00.01, vol.2. Agricultural economics research Institute (LEI), The Hague, the Netherlands. p. 115-127. ISBN 90-5242-563-9.

Poulsen, H.D., Børsting, C.F., Rom, H.B. and Sommer, S.G. 2001., Kvælstof, fosfor og kalium i husdyrgødning -normtal 2000. DJF rapport nr. 36. Husdyrbrug.

Winter, F., 2003. Empirisk model beregning af N_2O-emission og denitrifikation som funktion af jordtype og gødningsmængde. Unpublished note from Danmarks JordbrugsForskning. 10pp. (Empirical model of N_2O-emission and denitrifikation as a function of soil type and fertiliser application)

Weidema, B.P., Thodberg, L., Nielsen, A.H., Kristensen, I.S., Hermansen, J. & Hvid, S.K., 2002. Produktorienteret miljøindsats i landbrugets primærproduktion. Vurdering af virkemidler. Arbejdsrapport fra Miljøstyrelsen 19, 74 pp. http://www.mst.dk/udgiv/publikationer/2002/87-7972-131-1/html.

The ecoinvent database: use for the agri-food sector

<u>Nemecek, T.</u>, Erzinger, S. and Frischknecht, R.*

**Swiss Federal Research Station for Agroecology and Agriculture, FAL-Reckenholz, Reckenholzstr. 191, CH-8046 Zurich, Switzerland. Phone +41 1 3777254, fax +41 1 3777201, thomas.nemecek@fal.admin.ch.*

Swiss Federal Research Station for Agricultural Economics and Engineering, FAT-Taenikon, CH-8356 Ettenhausen, Switzerland.

ecoinvent Centre, c/o EMPA, Überlandstrasse 129, CH-8600 Dübendorf, Switzerland.

Abstract

Life cycle inventory (LCI) data are the basis of every LCA study and are very important for its quality. The ecoinvent database provides over 2500 LCI datasets from the areas energy systems, transportation, waste disposal, construction, chemicals, detergents, papers and agriculture with reference to Switzerland or Europe. It is useful to the agri-food sector by providing highly detailed data on agricultural plant products, infrastructure, means of production and processes and datasets from various other economic sectors having close relationships with the food industry. The data exchange format EcoSpold allows recording, documentation and exchange of LCI data in a standardised form.

Keywords: life cycle inventory, LCI, database, agricultural systems, agriculture, food industry

Introduction

Life cycle assessment (LCA) has proved to be a powerful tool for the environmental improvement of production processes in the agri-food sector (e.g. Anderson 2000). However, the increased use of the LCA method to analyse systems is hindered by the lack of agreement on the use of methods and by the poor availability of life cycle inventory (LCI) data.

The problem of availability of LCI data has various aspects:

- the data collection process is very time-consuming,
- production data are often confidential and not available to the public,
- LCI data are available in different, often incompatible formats (standardised formats like SPOLD are not widely used),
- data are collected for different regions and time periods, which leads to inconsistencies,
- the underlying standards and quality levels of the datasets are different.

To circumvent these difficulties, databases are needed that offer consistent data for many economic branches. Such a database has been built in the frame of the Swiss project ecoinvent 2000.

The ecoinvent database

The project ecoinvent 2000 was initiated by the ETH domain and Swiss Federal Offices to promote the life cycle approach in various economic sectors and to provide a basis for the "Integrated Product Policy". The goals of the project are:

- to harmonise the LCIs of the participating institutes on the basis of common quality guidelines and by using the common data exchange format EcoSpold (derived from the SPOLD99-standard),
- to update the data for the year 2000 and
- to allow access to the database via the internet and subsequently through different LCA software tools (see Jungbluth and Frischknecht 2003).

Twelve institutes have participated in the process of data acquisition and harmonisation. The database contains LCIs from the areas energy systems, transportation, waste disposal, construction, chemicals, detergents, papers and agriculture, which refer to the geographic context of Switzerland and/or Western Europe.

The main characteristics of the ecoinvent database are:

- The **documentation** is assured by **meta-data** in the database and by extended **reports**, provided on a CD-ROM.
- **Geographic reference**: each dataset refers to a specified country or region and to a defined process stage (e.g. "at plant", "at farm" or "at regional storehouse").
- **Common quality standards** were used to define and to document the datasets.
- The datasets were submitted to an **internal review process** prior to inclusion in the database.
- **Detailed recording of resources and emissions**: over 1000 resources and emissions belonging to 20 categories were distinguished (Table 1).
- **Land use** was accounted for by a newly developed scheme (see Jungbluth 2003).
- **Matrix calculation**: a mechanism of matrix inversion allows to solve the problem of mutual interdependencies of the datasets.
- The data exchange format **ECOSPOLD** is used to import and export the data and the meta-data (see Jungbluth and Frischknecht, 2003, see also Tables 3 and 4).
- The **uncertainty** of each figure is described by quantitative and qualitative indicators.

The ecoinvent database is accessible on the internet since September 2003. Further information is available on www.ecoinvent.ch.

LCI of agricultural systems in ecoinvent

Categories of emissions

The distinction of several subcategories in the ecoinvent database (Table 1) is useful for agricultural applications. The pollution of an industrial soil has certainly not the same impact and significance as the pollution of an agricultural soil which serves for the production of food. In the latter case, harmful substances can enter the human food chain and be ingested with consequent impacts on human health.

Table 1. Categories and subcategories of emissions and resources considered in ecoinvent. Only subcategories that contain data in ecoinvent data v1.0 are presented.

Category	SubCategory
air	low population density
	low population density, long-term
	lower stratosphere + upper troposphere
	high population density
	unspecified
soil	agricultural
	forestry
	industrial
	unspecified
water	ground-
	ground-, long-term
	lake
	ocean
	river
	unspecified
resource	in air
	biotic
	in ground
	land
	in water

The substances released during agricultural production are considered to be emitted to agricultural soil. Emissions in air from agricultural production are considered to go into “air, low population density”. On the other hand, the production processes for agricultural means of production like fertilisers and pesticides are assumed to take place in urban areas and consequently related emissions are counted as emissions into “air, high population density”. Low resp. high population density refer to the conditions in Central Europe. The distinction of these subcategories is relevant for impact categories like ozone formation or human toxicity.

All emissions to surface water from agricultural production are counted as emissions to “water, river” (e.g. P-emissions by run-off or erosion). As only Swiss agriculture was considered, emissions to sea water are irrelevant. Furthermore, since only a small share of the agricultural area is located in direct proximity of lakes, it was assumed that all emissions to surface water go into rivers. Leaching of nitrate and phosphorus are considered as emissions into ground water.

Field emissions

Direct field emissions are considered by means of models that partly use situation-specific parameters:

- Ammonia emissions are calculated with models described my Menzi *et al*. (1997). Constant release factors are applied in the case of mineral fertilisers. For slurry, liquid manure and solid manure the content of NH_4^+, the average monthly temperature and humidity and the quantity of manure spread per hectare are taken into consideration.
- Potential nitrate leaching is estimated on a monthly basis by accounting for N-mineralisation in the soil and the N-uptake by the vegetation (specific to each crop). If the mineralisation exceeds the uptake, nitrate leaching can potentially occur. In addition, a risk of nitrate leaching from fertiliser application during unfavourable periods is calculated, by taking the crop, the month of the application and the potential rooting depth into account (Oberholzer & Walther, 2001).
- Three paths of P-emissions to water are considered: run-off (as phosphate) and erosion (as phosphorus) to rivers and leaching to ground water (as phosphate). The category of land use, type of fertiliser, quantity of P spread and the characteristics and duration of the soil cover (for erosion) are considered. The model used is derived from Prasuhn & Grünig (2001).
- N_2O-emissions are estimated by using an adapted IPCC-method (Schmid *et al*., 2000). Indirect emissions from the conversion of NH_3 and NO_3^- into N_2O are considered in the inventory as well.
- Heavy metal emissions are assessed by a simple input-output balance, by taking all inputs (fertilisers, seed, pesticides) and outputs (products and straw) to resp. from the field caused by the farmer.
- Pesticide application are accounted for as emissions of the active ingredients in agricultural soil.

Resource use

CO_2 uptake by the vegetation is taken into account as well as the potential heat energy contained in the biomass. The biotic flows of CO_2 are clearly separated from the release of CO_2 during the combustion of fossil fuels. This lets the database user the opportunity either to consider biotic CO_2-flows (including also the CO2-release during the usage of the agricultural product) or not. The content of organic C in the soil is assumed as constant.

Land occupation and transformation is accounted for agricultural land as well as for non-agricultural areas (Jungbluth 2003).

Available datasets for agricultural systems and their use for the agri-food sector

The ecoinvent database provides three types of datasets useful for the agri-food sector:

1. **Datasets on agricultural products**. Data on plant products are included like the arable crop products wheat, rye, barley, silage and grain maize, potatoes, sugar and fodder beets, fava beans, soybeans, peas, sunflower and rape seed. For most crops datasets for integrated (IP, a production respecting a set of rules for environmental protection defined by the Swiss government) and organic production are present. For the cereals and rape seed an additional integrated extensive variant is calculated, corresponding to a crop without treatment by fungicides, insecticides and growth regulators, receiving extra subsidies in Switzerland. Three types of hay (intensive IP, intensive organic and extensive) are also included. The datasets refer to model crops, which have been defined on the basis of statistics (like FADN, import statistics, a.o.), recommendations, pilot farm networks and surveys of seed suppliers. The datasets were validated by a panel of experts. Table 2 shows an overview of the datasets of agricultural production in ecoinvent.
2. **Datasets on agricultural means of production**. The database contains a number of modules that allow the calculation of most systems in arable and fodder crop production and animal husbandry. LCAs of special productions like vineyards, vegetable and fruit production are partly possible. ecoinvent contains datasets on agricultural buildings, machinery, work and drying processes, fertilisers, pesticides, seed and animal feed (see Table 2).
3. **Background datasets for the food processing industry**. Furthermore, the database contains various modules that are required for LCA studies in the food sector: energy systems, transports, detergents, packaging materials, construction materials and processes as well as waste management.

Table 2. Overview of the datasets for agricultural systems available in the ecoinvent database.

	Subcategory	Number of modules	Example of inventories for the subcategories		
			Name	Location	Unit
Agricultural products	Plant production	59	potatoes organic, at farm	CH	kg
			rape seed extensive, at farm	CH	kg
			hay intensive IP, at farm	CH	kg
	Animal production	1	tallow, at plant	CH	kg
Agricultural inputs	Mineral fertilisers	24	lime, from carbonation, at regional storehouse	CH	kg
			ammonium nitrate, as N, at regional storehouse	RER	kg
			urea, as N, at regional storehouse	RER	kg
	Organic fertilisers	6	horn meal, at regional storehouse	CH	kg
	Pesticides	68	cyclic N-compounds, at regional storehouse	RER	kg
			[Sulfonyl]urea-compounds, at regional storehouse	CH	kg
			pesticide unspecified, at regional storehouse	CH	kg
	Seed	22	sugar beet seed IP, at regional storehouse	CH	kg
	Feed	10	wheat organic, at fodder mill	CH	kg
			wheat IP, at fodder mill	CH	kg
Infra-structure	Buildings	13	dried roughage store, air dried, solar	CH	kg
			label housing system, pig	CH	pig place
	Machinery	6	agricultural machinery, tillage, production	CH	kg
Operation of in-frastructure	Building usage	8	loose housing system, cattle, operation	CH	LU
			dried roughage store, air dried, solar, operation	CH	kg
	Machinery and equipment usage	33	haying, by rotary tedder	CH	ha
			tillage, ploughing	CH	ha
			milking	CH	kg
	Drying	4	grain drying, high temperature	CH	kg
	Total	254			

Examples from crop production

Table 3 shows an example of unit process inventory data and Table 4 of meta-data for the dataset “wheat, extensive CH”. The details of each dataset together with its description are available from the database.

Table 3. Example of a unit process inventory (multi-output process) in the ecoinvent database (wheat from extensive integrated production). *CH = Switzerland, RER = Europe, MA = Morocco, na = not considered.*

Unit process inventory for: 1 ha wheat extensive, CH									
					Uncertainty information			**Allocation**	
Input/ output	**Exchanges**	**Location/ Category**	**Unit**	**Value**	**Uncert Type**	**SD 95%**	Uncert Scores	**Wheat grains extensive, at farm CH (kg)**	**Wheat straw extensive, at farm CH (kg)**
Input	ammonium nitrate, as N, at regional storehouse	RER	kg	5.71E+01	1	1.07	(2,1,1,1,1,na)	92%	8%
Input	urea, as N, at regional storehouse	RER	kg	2.01E+01	1	1.07	(2,1,1,1,1,na)	92%	8%
Input	diammonium phosphate, as N, at regional storehouse	RER	kg	5.10+E00	1	1.07	(2,1,1,1,1,na)	92%	8%
Input	calcium ammonium nitrate, as N, at regional storehouse	RER	kg	2.87E+01	1	1.07	(2,1,1,1,1,na)	92%	8%
Input	ammonium sulphate, as N, at regional storehouse	RER	kg	4.32E+00	1	1.07	(2,1,1,1,1,na)	92%	8%
Input	triple super phosphate, as P205, at regional storehouse	RER	kg	1.91E+01	1	1.07	(2,1,1,1,1,na)	92%	8%
Input	single super phosphate, as P205, at regional storehouse	RER	kg	7.71E-01	1	1.07	(2,1,1,1,1,na)	92%	8%
Input	diammonium phosphate, as P205, at regional storehouse	RER	kg	1.30E+01	1	1.07	(2,1,1,1,1,na)	92%	8%
Input	phosphate rock, as P205, beneficiated, dry, at plant	MA	kg	1.12E+01	1	1.07	(2,1,1,1,1,na)	92%	8%
Input	thomas meal, as P205, at regional storehouse	RER	kg	2.42E+00	1	1.07	(2,1,1,1,1,na)	92%	8%
Input	potassium chloride, as K20, at regional storehouse	RER	kg	1.56E+01	1	1.07	(2,1,1,1,1,na)	92%	8%
Input	potassium sulphate, as K20, at regional storehouse	RER	kg	1.03E+00	1	1.07	(2,1,1,1,1,na)	92%	8%
Input	tillage, ploughing	CH	ha	1.00E+00	1	1.07	(2,1,1,1,1,na)	92%	8%
Input	tillage, harrowing, by spring tine harrow	CH	ha	2.00E+00	1	1.07	(2,1,1,1,1,na)	92%	8%
Input	tillage, currying, by weeder	CH	ha	1.00E+00	1	1.07	(2,1,1,1,1,na)	92%	8%
Input	sowing	CH	ha	1.00E+00	1	1.07	(2,1,1,1,1,na)	92%	8%
Input	fertilising, by broadcaster	CH	ha	4.00E+00	1	1.07	(2,1,1,1,1,na)	92%	8%
Input	solid manure loading and spreading, by hydraulic loader and spreader	CH	kg	9.62E+02	1	1.07	(2,1,1,1,1,na)	92%	8%
Input	slurry spreading, by vacuum tanker	CH	m3	8.25E+00	1	1.07	(2,1,1,1,1,na)	92%	8%
Input	application of plant protection products, by field sprayer	CH	ha	1.00E+00	1	1.13	(2,2,3,1,1,na)	92%	8%
Input	combine harvesting	CH	ha	1.00E+00	1	1.07	(2,1,1,1,1,na)	92%	8%
Input	transport, tractor and trailer	CH	tkm	1.39E-02	1	1.07	(2,1,1,1,1,na)	100%	
Input	grain drying, low temperature	CH	kg	6.39E+01	1	1.07	(2,1,1,1,1,na)	100%	
Input	baling	CH	unit	4.70E+00	1	1.07	(2,1,1,1,1,na)		100%
Input	loading bales	CH	unit	2.05E+01	1	1.07	(2,1,1,1,1,na)		100%
Input	tillage, cultivating, chiselling	CH	ha	1.00E+00	1	1.07	(2,1,1,1,1,na)	92%	8%
Input	nitrile-compounds, at regional storehouse	CH	kg	2.00E-01	1	1.13	(2,2,3,1,1,na)	92%	8%
Input	phenoxy-compounds, at regional storehouse	CH	kg	3.50E-01	1	1.13	(2,2,3,1,1,na)	92%	8%

Input	(sulfonyl)lurea-compounds, at regional storehouse	CH	kg	7.20E-01	1	1.13	(2,2,3,1,1,na)	92%	8%
Input	wheat seed IP, at regional storehouse	CH	kg	1.80E+02	1	1.07	(2,1,1,1,1,na)	92%	8%
Input	transport, freight, rail	CH	tkm	5.87E+01	1	2.71	(4,5,na,na,na,na)	92%	8%
Input	transport, lorry 28t	CH	tkm	4.68E+01	1	2.71	(4,5,na,na,na,na)	92%	8%
Input	transport, van <3,5t	CH	tkm	2.74E+00	1	2.71	(4,5,na,na,na,na)	92%	8%
Input	transport, lorry 40t	CH	tkm	7.00E+00	1	2.71	(4,5,na,na,na,na)	92%	8%
Input	transport, barge	RER	tkm	4.66E+02	1	2.71	(4,5,na,na,na,na)	92%	8%
Input	Occupation, arable, non-irigated	resource/land	m2a	7.94E+03	1	1.77	(2,1,1,1,1,na)	92%	8%
Input	Transformation, from pasture and meadow, intensive	resource/land	m2	2.90E+03	1	2.67	(2,1,1,1,1,na)	92%	8%
Input	Transformation, from arable, non-irigated	resource/land	m2	7.10E+03	1	2.67	(2,1,1,1,1,na)	92%	8%
Input	Transformation, to arable, non-irigated	resource/land	m2	1.00E+04	1	2.67	(2,1,1,1,1,na)	92%	8%
Input	Carbon dioxide, in air	resource/in air	kg	1.16E+04	1	1.07	(2,2,1,1,1,na)	61%	39%
Input	Energy, gross calorific value, in biomass	resource/biotic	MJ	1.40E+05	1	1.07	(2,2,1,1,1,na)	59%	41%
Output	Dinitrogen monoxide	air/low population density	ykg	4.30E+00	1	1.61	(2,2,1,1,1,na)	92%	8%
Output	Nitrogen oxides	air/low population density	ykg	9.03E-01	1	1.61	(2,2,1,1,1,na)	92%	8%
Output	Ammonia	air/low population density	ykg	1.06E+01	1	1.30	(2,2,1,1,1,na)	92%	8%
Output	Nitrate	water/ground-	kg	1.73E+02	1	1.77	(2,2,1,1,1,na)	92%	8%
Output	Phosphate	water/ground-	kg	1.85E-01	1	1.77	(2,2,1,1,1,na)	92%	8%
Output	Phosporus	water/river	kg	2.58E-01	1	1.77	(2,2,1,1,1,na)	92%	8%
Output	Phosphate	water/river	kg	5.64E-01	1	1.77	(2,2,1,1,1,na)	92%	8%
Output	Cadmium	soil/agricultural	kg	2.72E-03	1	1.77	(2,2,1,1,1,na)	42%	58%
Output	Copper	soil/agricultural	kg	-1.82E-02	1	1.77	(2,2,1,1,1,na)	55%	45%
Output	Lead	soil/agricultural	kg	-9.56E-03	1	1.77	(2,2,1,1,1,na)	5%	95%
Output	Zinc	soil/agricultural	kg	-1.39E-01	1	1.77	(2,2,1,1,1,na)	79%	21%
Output	Nickel	soil/agricultural	kg	6.77E-03	1	1.77	(2,2,1,1,1,na)	47%	53%
Output	Chromium	soil/agricultural	kg	5.52E-02	1	1.77	(2,2,1,1,1,na)	46%	54%
Output	Mercury	soil/agricultural	kg	-1.39E-04	1	1.77	(2,2,1,1,1,na)	18%	82%
Output	Difenoconazole	soil/agricultural	kg	1.80E-02	1	1.32	(2,2,3,1,1,na)	92%	8%
Output	Ioxynil	soil/agricultural	kg	2.00E-01	1	1.32	(2,2,3,1,1,na)	92%	8%
Output	Isoproturon	soil/agricultural	kg	7.20E-01	1	1.32	(2,2,3,1,1,na)	92%	8%
Output	Mecoprop-P	soil/agricultural	kg	3.50E-01	1	1.32	(2,2,3,1,1,na)	92%	8%
Output	wheat grains extensive, at farm	CH	kg	5.37E+03				100%	
Output	wheat straw extensive, at farm	CH	kg	3.27E+03					100%

Table 4. Example of meta-data for "wheat extensive, CH" (extract).

Type	**Field name**	**Field contents**
ReferenceFunction	Name	Wheat extensive
Geography	Location	CH
ReferenceFunction	InfrastructureProcess	0 (=no)
	Unit	Ha
	Amount	1
	IncludedProcesses	The inventory includes the processes of soil cultivation, sowing, weed control, fertilisation, pest and pathogen control, harvest and drying of the grains. Machine infrastructure and a seed for machine sheltering is included. Inputs of fertilisers, pesticides and seed as well as their transports to the farm are considered. The direct emissions on the field are also included. The system boundary is at the farm gate.
	LocalName	Weizen Extenso
	GeneralComment	Inventory refers to the production of 1 kg wheat extensive grains respectively straw, both with a moisture content of 15%. The multioutput-process 'wheat extensive' delivers the co-products 'wheat grains extensive, at farm' and 'wheat straw extensive, at farm'. Economic allocation with allocation factor of 92.5% to grains (exceptions see report).
	Category	agricultural production
	SubCategory	plant production
TimePeriod	StartDate	1996
	EndDate	1999
	DataValidFor EntirePeriod	1 (=yes)
	OtherPeriodText	The yield data have been collected for the years 1996-1999.
Geography	Text	Refers to an average production in the Swiss lowlands.
Technology	Text	Integrated production with extensive plant protection (no fungicides, insecticides and growth regulators)
Representativeness	Percent	86
	ProductionVolume	CH prod of wheat: 561000t in 2000. % refers to fract of tot area in lowlands
	SamplingProcedure	Data were compiled from statistics, pilot network, fertilising recommendations, documents from extension services, information provided by retailers and expert knowledge. The production data were verified and adjusted by a group of experts.

Figure 1 shows an example of calculated results for several arable crops. The cumulative energy demand (CED) is determined by the use of machines (mechanisation), the application of mineral fertilisers and by grain drying. For mechanisation the processes of soil cultivation/seedbed preparation and harvest are most relevant. The production of mineral fertilisers – especially nitrogen fertilisers – is highly demanding on energy resources. Mineral fertilisers have the highest share in cereals and oil crops. Drying is very important in grain maize, followed by oil crops, grain legumes and cereals. This shows the importance of harvesting in dry conditions. Pesticide production has only little relevance for energy use.

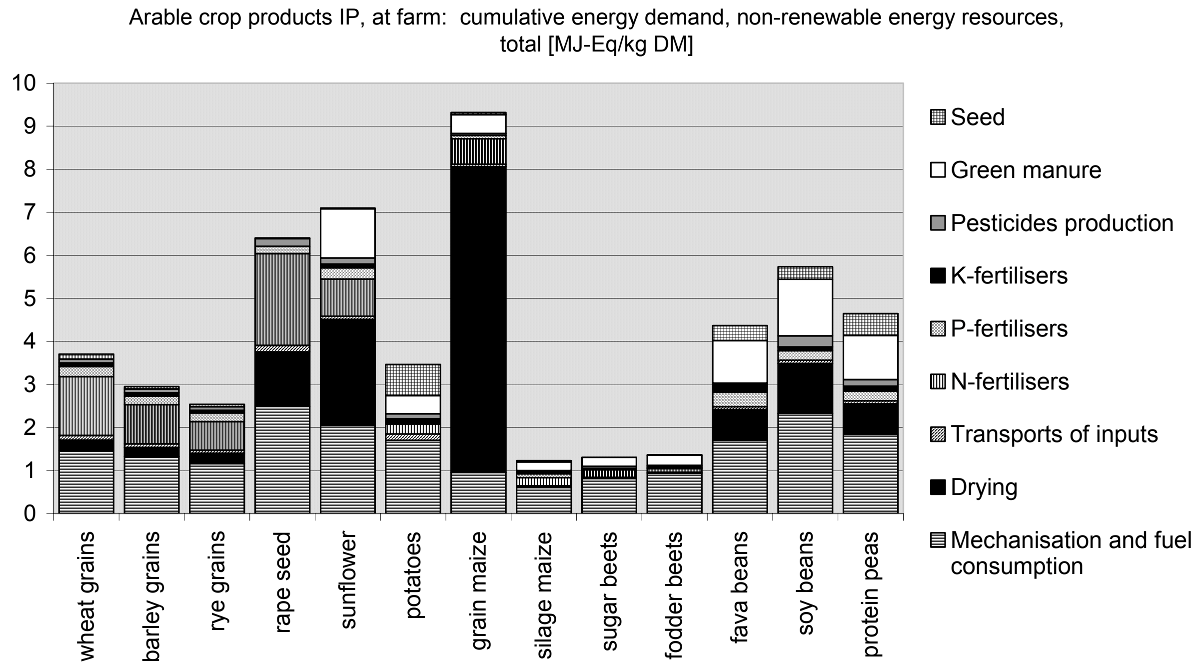


Figure 1. Cumulative energy demand of non-renewable energy resources for agricultural crops from integrated production per kg dry matter (DM).

Discussion and conclusion

The ecoinvent database provides datasets that allow the calculation of LCA for agricultural production and food processing. The database offers not only data on the agricultural sector, but also on the most important economic sectors. It is detailed and accompanied by an extensive documentation.

The database contains also minima and maxima (confidence limits) for each figure. This allows to estimate the uncertainty of the information. However, these indications are rather rough and would profit from methodical developments in the future. In addition, the parameters are varied independently in the Monte-Carlo simulations, although they are not independent in reality.

Further developments of the database can be the extension to other economic sectors not yet covered and to countries outside Europe. For agriculture in particular, the database would profit from including animal products and datasets from other European countries.

The ecoinvent contains a large number of inventory datasets containing detailed data on resources and emissions for 20 environmental subcategories. Most current LCIA methods do not make use of this information. We encourage scientists who are developing LCIA methods to consider the structure now available and to refine their characterisations factors for the different subcategories. This will contribute to an increased reliability and credibility of LCA.

The EcoSpold format allows to exchange unit-process inventory data, LCI results and also LCIA methods and results in the same format. Since an EcoSpold-interface is implemented in the most important LCA-software tools, this is a great progress towards better data exchange and more efficient LCA studies. It is expected that ecoinvent should boost the application of the LCA method in the agri-food sector.

References

Andersson, K., 2000. LCA of food products and production systems. Int. J. LCA, **5** (4): 239-248.

Jungbluth, N., 2003. Land occupation and transformation in life cycle inventories. Paper presented at the 4th International Conference on: Life Cycle Assessment in the Agri-food sector, Horsens, Denmark, October 6-8, 2003.

Jungbluth, N. and Frischknecht R., 2003. Life cycle inventory modelling in the Swiss ecoinvent database. Poster presented at the 4th International Conference on: Life Cycle Assessment in the Agri-food sector, Horsens, Denmark, October 6-8, 2003.

Menzi, H., Frick, R. & Kaufmann, R., 1997. Ammoniak-Emissionen in der Schweiz: Ausmass und technische Beurteilung des Reduktionspotentials. Schriftenreihe der FAL 26, Zürich, 124 p.

Nemecek, T., Heil, A., Huguenin, O., Meier, S., Erzinger, S., Blaser, S., Dux, D. and Zimmermann A., 2003. Life Cycle Inventories of Agricultural Production Systems. Final report ecoinvent 2000 No. 15. FAL Reckenholz, FAT Tänikon, Swiss Centre for Life Cycle Inventories, Duebendorf, CH, Online-Version under: www.ecoinvent.ch.

Oberholzer, H.R. & Walther, U. (2001) Modell zur Abschätzung des Nitratauswaschungsspotenzials. In: Neue Erkenntnisse zu Stickstoffflüssen im Ackerbau, FAL-Tagung 6.4.2001, 6p.

Prasuhn V. & Grünig K. (2001) Evaluation der Ökomassnahmen Phosphorbelastung der Oberflächengewässer durch Bodenerosion. Schriftenreihe der FAL 37, 151p.

Schmid, M., Neftel, A. & Fuhrer, J., 2000. Lachgasemissionen aus der Schweizer Landwirtschaft. FAL, Schriftenreihe der FAL 33, 131 p.

Life Cycle Inventory of the Galician dairy sector

Hospido, A., Moreira, M.T. and Feijoo, G.*

Department of Chemical Engineering. University of Santiago de Compostela. 15782 – Santiago de Compostela (Spain). T: +34981563100, ext.16020 // F: +34981547168 // e-mail: ahospido@usc.es

Abstract

Dairy industry has been extensively studied from Life Cycle Assessment (LCA) in many European countries, nevertheless, concerning LCA in Spain, little work has been reported, and a global and reliable inventory is still lacking. In this work particular attention has been paid to the Life Cycle Inventory (LCI) of this sector by performing an exhausted fieldwork considering the contribution of several subsystems: farms, fodder factories, dairies and the manufacture of tetrabrik containers. For each subsystem, average data as well as associated standard deviations are presented. The majority of these data suffer from a high variability, which means that representative production and processing scheme can be difficult to establish.

Key words: life cycle inventory, dairy, farm, milk, and fodder

Background

In April 2000, the European Commission published an extensive report concerning the environmental impact of the dairy production in the European Union (CEAS Consultants Ltd, 2000), where a classification of countries according to herd size was presented. Spain was included on the third group characterized by a huge number of many small farms. Most of the LCA studies on milk production were carried out for countries belonging to the second one (Berlin, 2002; Cederberg and Mattson, 2000; Haas et al., 2001; Høgaas, 2002), but in the case of the third group, a reliable inventory for Life Cycle Analysis was still lacking.

In Galicia, a Spanish region with an important milk production, statistical data indicate that 75% of its farms have less than 10 cows, while the remaining 25% have encountered an industrialisation and modernisation process with a superior surface and herd size, which correspond to a renovated picture of the rural area.

Moreover, since mid-80´s dairy facilities in Galicia have suffered an important recession, a high number of farms have been obliged to close: 1,159 on year 2,001 and 1,435 on year 2,002, this rising tendency being observed during last years. In fact, the number of farms in Galicia has decreased from 109,284 farms in 1,982 to 24,910 at the end of 2,002. The earliest farms to undergo these consequences are the smaller ones because they cannot be competitive so the farms that have a significant contribution in the productive framework are the bigger ones. For this reason, three of them were selected for this study.

Milk processing is a more uniform process as the technology and facilities are common for the majority of the dairies. Three dairies, which an important market quota, have been chosen to be inventoried.

Method

Goal and Scope Definition

The objective of this study is to examine the total life cycle of production and processing of milk in order to quantify the potential environmental impact. Three dairies and three farms have been selected as representative of Galician milk industry to define both production and processing scenarios. Additionally, other relevant subsystems have been identified and studied separately: fodder production (also called concentrate mixture) and packaging, the former by considering two important fodder factories and the latter by public report data. Our main focus of attention is LCI, the second phase at LCA methodology. This stage requires the collection of extensive data on the physical inputs and outputs of the processes and related procedures under evaluation. As it was mentioned before, these data were compiled on a case-by-case basis bearing in mind the importance of data with high quality and reliability.

Functional unit

The functional unit (FU) selected is 1 litre of packaged liquid milk, ready to be delivered, which corresponds to the standard and more widespread final presentation of milk at Galician and Spanish homes.

System boundaries

The life cycle of milk production included in this analysis is shown in Figure 1, where the following hierarchy has been established:

- First level: Main stages of global process.
- Second level: Inputs that have an industrial process associated and have been analysed in detail.
- Third level: Inputs taken from LCA databases.

Apart from the main product, other outputs of the system such as co-products, waste and emissions to water, air or soil, are also included in the inventory. The most often omitted subsystem is the consumer phase and it has not been considered in the present study.

Allocation rules

During the performance of LCA, allocation problems arise when the life cycle of different products are connected. In other works (Høgaas, 2002), a cow is defined as a multifunctional production system, supplying several products: milk, meat, skin and manure. The allocation rule applied by Høgaas for the two main products (milk and meat) was based on the biological demand of fodder while the other two products were not considered: the manure was used as a fertiliser at the farm and it was not looked upon as a product, and the skin of the cows was omitted due to its much lower importance (economic and mass). Although the farms evaluated were specialised in milk production, the study of economic revenue per cow based on historical Galician market prices from years 2000-2002 for milk and meat has entailed the following distribution of the associated economical benefits: 87% for milk and 13% for meat. This figure is important and therefore economic allocation was used for raw milk production in our study.

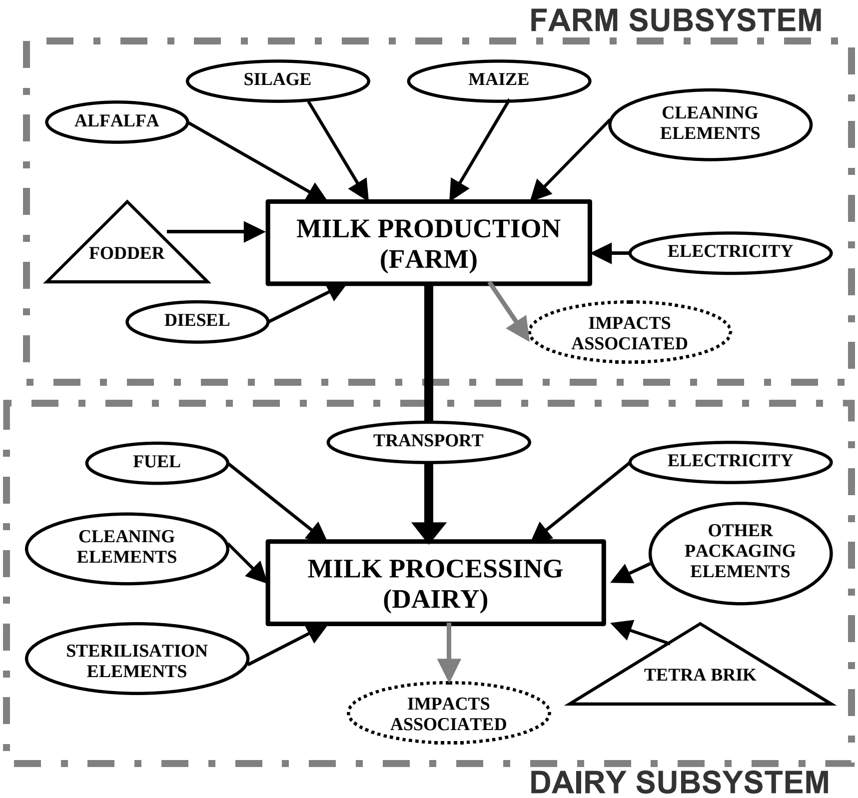


Figure 1. Schematic flow chart of the life cycle of milk. The main stages of the process are represented in blocks; Inputs associated with industrial processes in triangles; Inputs from LCA databases in continuous circles; impacts associated in discontinuous circles.

At the milk processing stage, the dairies chosen are basically mono-functional and the cream obtained as a co-product represents less than 2.5% of the total annual production, consequently, allocation rules considering cream were not applied. During fodder production and tetrabrik manufacture process, mass criteria have been chosen depending on the distribution of final products.

Data Collection

To assess the most accurate environmental impacts associated to Galician milk production, we mainly considered data from Galician industries. Real data from farms, fodder factories and dairies were collected in consecutive periods during the last three years. In particular, elements coming from the agricultural subsystem (maize, alfalfa and silage) have been quantified and data corresponding to their harvest belong to a comprehensive study made in Catalonia (Milá et al., 1998).

There are other types of data whose production systems are not present in Galicia and they have been obtained from companies from other regions. For instance, tetrabrik containers are only manufactured in a factory located in Arganda del Rey (Madrid) and its inventory is obtained from several reports (Tetrapack Spain, 1999, 2000 and 2001). In relation to the electricity production profile, an electrical percentage distribution according to data from the Institute for Diversification and Energy Saving (IDAE, 2002) has been used: 35.8% of the electricity is produced from coal, 27.6% is nuclear, 13.9% is hydroelectric, 9.9% is obtained from oil power plants, 9.7% from gas power plants, 2.2% from wind power plants, 0.6% from waste use and 0.3% from biomass use. However, due to the non-availability of data quantifying the environmental burdens associated to the different ways to produce electricity in Spain, we chose data from the database BUWAL 250 (SAEFL, 1998).

Results

LCI at Farms

Three well-managed farms were inventoried (Table 1). Their renewed facilities, consisting of an automatic milking system with recollection pipes and a storage tank, are a good example of the modernisation philosophy that is in practice nowadays.

Table 1. General data about the farms inventoried.

	Farm A	Farm B	Farm C
Location	Portomarin (Lugo)	Carballedo (Lugo)	Sarria (Lugo)
Operative since	1997	1996	1994
Annual Production (L)	228,286 (year 2000)	353,725 (year 2001)	427,050 (year 2002)
Average Size (animals)	50	60	67
Dairy cows	23	38	50
Suckler cows	27	22	17
Average yield (L/cow/year)	9,925	9,309	8,541

To obtain the annual milk production for each farm, a daily measurement took place. Then, this yearly production is handled to obtain the inventory data on the basis of the chosen functional unit considering annual consumption. Figure 2 shows bimonthly electricity spending form Farms B and C that were measured to get the annual consumption.

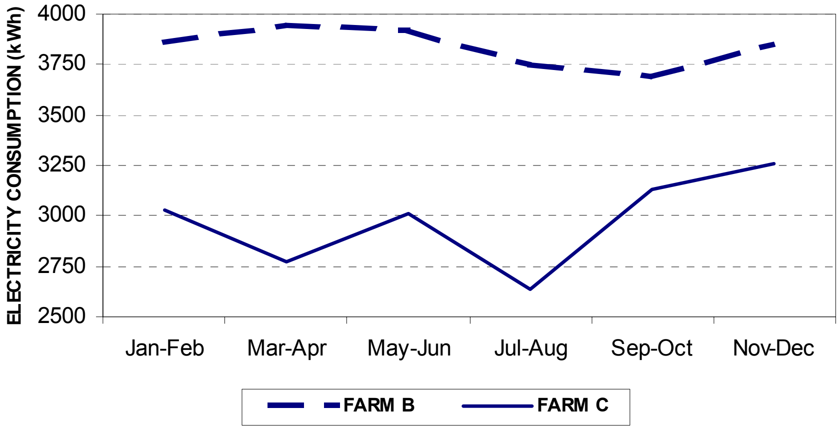


Figure 2. Bimonthly variation of electricity consumption at Farms B and C.

Table 2 shows the results of the fieldwork developed there; the average values as well as the standard deviations are presented. Data have been mainly quantified according to the annual consumption of each element and the opinion of the farmers.

Table 2. Inventory Data of the Farm (FU = 1 L of raw milk).

INPUTS			
From the TECHNOSPHERE		From the NATURE	
Materials and fuels		Raw materials	
1. Food		1. Water	4.20 ± 0.25 L
1a. Maize	482.9 ± 191.1 g		
1b. Fodder	424.0 ± 36.7 g		
1c. Silage	551.6 ± 208.1 g		
1d. Alfalfa	145.0 ± 88.7 g		
2. Alkaline detergent	0.78 ± 0.06 mL		
3. Acid solution	0.11 ± 0.06 mL		
4. Disinfectant	0.83 ± 0.62 mL		
5. Paper	0.418 g		
6. Linchpin	$9.00 \cdot 10^{-5}$ units		
7. Diesel	4.25 ± 0.10 mL		
Electricity			
1. Electricity	50.25 ± 12.96 Wh		
OUTPUTS			
To the TECHNOSPHERE		To the NATURE	
Products and co-products		Emissions to air	
1. Raw milk	1 L	1. Methane *	13.01 ± 0.99 g
Waste for treatment		Emissions to water	
1. Urban Solid Waste	0.28 g	1. Wastewater **	1.30 ± 0.10 L
		COD	8.22 g L^{-1}
		TSS	2.70 g L^{-1}

* According to available data (EPA, 2002), an adult cow emits 120 kg of methane per year due to enteric fermentation.
** COD = Chemical Organic Demand // TSS = Total Suspended Solids

Regarding food components, maize and silage are produced on each farm (internal elements) whereas fodder and alfalfa are considered external elements that have to be exported from outside.

LCI at Fodder Factories

Maize, fodder, silage and alfalfa are the main components in animal food, and their final mixture has to attain acceptable levels of protein and energy supplement (between 15.5-18.5% protein and 1.70-1.72 Mcal kg^{-1} of dry material).

Fodder plays an important role in animal food and this subsystem has been studied in detail and a life cycle inventory of this manufacture process has been carried out. The inventory data proceeded from two factories sited in Galicia (Table 3). The first one has an annual production of over 100,000 tonnes and an animal distribution of 60% for cattle, 35% for pig and 5% for other animals. The second one has an annual production of 90,000 tonnes, which is distributed in the following percentages: 90% for cattle destined to milk production and 10% for cattle belonging to rural families destined to their own milk consumption. The distribution of the production in terms of areas of influence is the following: 98% of total production is delivered within 30-40 km and 2% goes to longer distances (100 km).

LCI at Dairies

All the Galician dairies have a similar technology and a comparable size, so the dairy inventory data were calculated on the basis of their annual production for the years 2001 and 2002: around 200 millions litres. The products produced are packaged liquid milk: 71%, whole; 18%, semi-skimmed and 11%, skimmed milk. Cream is obtained as a co-product and it is sold for further processing in other factories.

Transport by isothermal trucks from farms to dairies has been included and studied in detail, identifying all the routes associated to each factory as well as their distance and amount of milk collected.

Table 4 displays the inventory data for Dairy Subsystem.

Table 3. Inventory Data of the Fodder Factory (FU = 1 kg of fodder).

INPUTS			
From the TECHNOSPHERE		From the NATURE	
Materials and fuels		Raw materials	
1. Raw materials		1. Water	66.62 mL
1a. Maize	165.8 ± 34.2 g		
1b. Barley	189.2 ± 15.9 g		
1c. Wheat	47.7 ± 7.7 g		
1d. Rye	82.22 g		
1e. Soy bean	143.7 ± 10.4 g		
1f. Soy shell	19.4 ± 0.6 g		
1g. Gluten	104.8 ± 15.2 g		
1h. Cotton seed	34.7 ± 5.3 g		
1i. Molasses	22.1 ± 1.3 g		
1j. Calcium carbonate.	13.2 ± 3.9 g		
1k. Phosphate carbonate.	4.6 ± 2.1 g		
1l. Alfalfa	3.6 ± 1.7 g		
2. Paper bags	1.39 ± 0.03 g		
Electricity			
1. Electricity	49.1 ± 3.3 Wh		
OUTPUTS			
To the TECHNOSPHERE		To the NATURE	
Products and co-products			
1. Fodder	1 kg		
Waste for treatment			
1. Urban Solid Waste	0.11 kg		
2. Oil	0.08 g		

Discussion

Average data as well as the associated standard deviations have been presented separately for each subsystem. As it can be observed, the majority of these data suffer from a high variability (defined as the ratio between the standard deviation and the mean), which means that representative production and processing scheme can be difficult to establish. For instance, all the elements included on the food ration, with the exception of fodder or concentrate feed, undergo variability around 50%.

The next step will be the evaluation of this inventory in order to identify the most pollutant subsystems along the process. In this study, focus has been paid on the LCI so this will be beyond the scope of the study.

Table 4. Inventory Data of the Dairy (FU = 1 L of packaged milk).

INPUTS			
From the TECHNOSPHERE		From the NATURE	
Materials and fuels		Raw materials	
1. Raw milk	1.11 ± 0.09 L	1. Water	3.18 ± 1.74 L
2. Tetra-brik	1.01 ± 0.01 u		
3. Cardboard	12.26 ± 6.42 g		
4. Film	1.12 ± 1.33 g		
6. Hydrogen peroxide	0.69 ± 0.34 g		
7. Nitric acid	1.12 ± 1.04 g		
8. Sodium hydroxide	1.95 ± 0.45 g		
9. Fuel	8.06 ± 2.32 g		
Transport			
1. Truck 40 ton	281 ± 104 kgkm		
Electricity			
1. Electricity	46.12 ± 10.90 Wh		
OUTPUTS			
To the TECHNOSPHERE		To the NATURE	
Products and co-products		Solid emissions	
1. Packaged milk	1 L	1. Combustion waste	2.155 g
2. Cream	46.42 ± 34.16 g	Emissions to air	
Waste for treatment		1. SO_2	0.20 ± 0.01 g
1. Cardboard	0.34 ± 0.05 g	2. NO_x	6.83 ± 4.75 g
2. Film	0.785 g	3. CO	2.14 ± 2.39 g
3. Used oil	0.04 g	Emissions to water	
4. Used oil filters	10^{-7} units	1. Wastewater	1.06 ± 1.25 L
5. Used Tetra-brik	0.008 ± 0.006 u	Emissions to soil	
		1. Sludge	40.46 ± 17.74 g

Future Outlook

The research planned for the future will continue in several directions:

- In order to achieve a higher quality of data at farm level, we are trying to proceed with a more complete inventory including new production systems for each stage. For instance, we are carrying out the inventory of two additional farms, which are examples of completely different herd sizes (one farm of 20 cows and the other of 170 heads) in order to identify the influence of extreme cases on the environmental impact associated. At the moment those inventories are not finished so the conclusions about this aspect are not still available. Regarding dairy level and bearing in mind the average technology and specialising degree at Galician factories, this influence is supposed to be not significant and this is the reason why we are concentrated on farm level.

- We would like to get a more comprehensive knowledge about the influence of cereals on the fodder factories. We have used data from international databases on the basis of the assumed country of origin for each one. The next step to check the reliability of this assumption is the comparison with the data handled by the major farm cooperative that manages imports and supplies raw cereals to an important number of the fodder factories in Galicia.
- Another point of attention will be the consideration of the named Ecological Milk, which is commercialised in our region since 2002. Nowadays, only one company is processing this ecological product and its market share is still small, but bearing in mind the social growing environment awareness, this product is supposed to have a relevant contribution in a near future.

References

Berlin, J., 2002. Environmental Life Cycle Assessment (LCA) of Semi-Hard Cheese. *International Dairy Journal* **12**, 939-953.

CEAS Consultants (Wye) Ltd and The European Forum on Nature Conservation and Pastoralism, 2000. The environmental impact of dairy production in the EU, CEAS 1779/BDB, Brussels, Belgium. 176 pp. Available at http://europa.eu.int/comm/environment/agriculture/pdf/dairy.pdf

Cederberg, C. and Mattson, B., 2000. Life cycle assessment of milk production - a comparison of conventional and organic farming. *Journal of Cleaner Production* **8**, 49-60.

EPA, 2002. U.S. Environmental Protection Agency, www.epa.gov.

Haas, G., Wetterich, F. and Köpke, U., 2001. Comparing intensive, extensified and organic grassland farming in southern Germany by process life cycle assessment. *Agriculture, Ecosystems & Environment* **83**, 43-53.

Høgaas, M.E., 2002. Life Cycle Assessment of Industrial Milk Production, Chalmers University of Technology, Gothenburg, Sweden. 53 pp.

IDAE, 2002. Instituto para la Diversificación y Ahorro de la Energía (in Spanish), www.idae.es.

Milá, L., Rieradevall, J., Domènech, X., Fullana, P. and Puig, R., 1998. Anàlisi del Cicle de Vida de la Pell (in Catalonian), Bellaterra, Spain.191 pp.

SAEFL, 1998. Life Cycle Inventories for Packagings, Environmental Serie No. 250/I and 250/II, Berne, Switzerland. 552 pp.

Tetrapack Spain, 1999. Declaración Ambiental 1998 (in Spanish). 10 pp.

Tetrapack Spain, 2000. Declaración Ambiental 1999 (in Spanish). 10 pp.

Tetrapack Spain, 2001. Informe de Sostenibilidad 2000 (in Spanish). 52 pp. Available at http://www.tetrapak.es/sostenibilidad/index.htm

Identification of processes affected by a marginal change in demand for food products - two examples on Danish pigs and cheese

Anne Merete Nielsen*, Per H. Nielsen[1], Jørgen Dejgård Jensen[2], Martin Andersen[2] and Bo P. Weidema[1].
*) 2.-0 LCA consultants, Borgergade 6, 1., DK-1300 Copenhagen. Tel. +45 333 22 8 22. Fax: +45 339****. AMN@LCA.DK. 1) 2.-0 LCA consultants, Borgergade 6, 1., DK-1300 Copenhagen. 2) Danish Research Institute of Food Economics, Rolighedsvej 25, DK-1958 Frederiksberg C.

Abstract
In environmental assessments of products, co-products should be dealt with by use of system expansion. This theoretical consensus has existed for some time. However, till now still many environmental assessments are based on economically or mass allocated data. The LCAfood database is the first consistent model of Danish food-production, with a widespread use of system expansion. Using quantified as well as more qualitative knowledge on market structures and production economics, the affected processes are identified for a range of basic food products, in agriculture as well as in food processing industry. It is crucial to identify which technologies are affected by a product demand prior to data collection, as the work can be focused on the most important processes, and the explanatory power of the environmental assessment can be maximised.

Keywords: environmental assessment, milk production, system expansion, milk-quotas

Background

Agriculture has significant contribution to many negative environmental impacts (see eg. Kramer, 2000), and product-oriented environmental assessment is a promising approach to achieve new insights on how to decrease environmental impacts from food consumption. Because of agriculture's high degree of co-productions due to crop rotations and linked production of plants and animals, it has not been straightforward to find out how to assess the impacts of the individual products. The theoretical consensus has existed for some time that co-products should be dealt with by use of system expansion (Guinée et al., 2001), and some basic principles for the implementation are available (Weidema et al. 1999, and Weidema, 2001). However, till now most environmental assessments of agricultural products are based on economically or mass allocated data.

In system expansion, the main-product for which the production is optimised should be identified, and the credit of the by-products should be given by subtracting the processes, which they affect, e.g. an alternative production of a similar product. A key-point in the application of system expansion is therefore the identification of the affected producer/technology.

To identify affected processes, it is necessary to model consequences in society, and treat the dynamics of human behaviour stringent and exact, even though they lie outside the field of natural science, which is the base of most practitioners of environmental assessments. Luckily other sciences offer the models for these dynamics, such as economy, marketing and politology. A limiting factor for inclusion of knowledge from these sources can be the perception of "objectiveness" in natural science. Logics and models from these other sciences are often perceived as subjective.

For years, the use of average versus marginal data has been discussed in the field of life-cycle assessment (LCA). However, in economic analysis it is a given that analysis made to support decisions on the most efficient use of resources should be based upon marginal data, i.e. knowledge on the consequences of the change under study, all other things being equal (see e.g. Hardwick et al., 1990). Now the theoretical consensus seems to be formed among LCA-practitioners as well: if the assessment shall predict the consequences of an action, data must come from the processes, which are actually affected.

The LCAfood database (Nielsen et al, 2003) is the first consistent model of Danish food-production, with a widespread use of system expansion and application of marginal data. In this paper two examples of system expansion with affected processes are presented.

Framework for identification of affected processes

Weidema et al. (1999) provided the first applicable guide to identification of affected processes in environmental assessments. Rephrasing, but following the string of logic, our identification of affected producers has gone through two kinds of questions.

1. Is the supplying market increasing or decreasing, and are any of the supplying technologies constrained?

This question addresses the nature of the market, which delivers the products, and is necessary to identify which of the possible suppliers are most likely to be affected by a small change in demand. If the market is increasing, the general trend will be that new productions are being established, with the most competitive technologies. A little change in demand will therefore affect these technologies. If the market is decreasing on the other hand, the general trend will be, that the least competitive producers are stopping their production. A little extra demand will therefore make them stay longer or shorter on the market.

Some producers can be constrained by technical limits (e.g. availability of land), legislation (e.g. limits to application of fertiliser) or other limits.

2. Which supplier is most likely to be affected by a small change in demand?

To answer this, one must list the technologies, which are technically capable of delivering the relevant product (as identified in question 1) and identify which of them is a) unconstrained and b) most respectively least competitive (for increasing respectively decreasing market).

To answer this, the total production-economical costs and benefits of the technologies should be estimated more or less quantitatively.

Processes affected by demand for pork

The nature of the market for agricultural products

The majority of Danish pigs are produced at highly specialised farms, but productions also exist at more diverse farms, such as sugar beet or potato-producing farms. Pork is traded on a global, increasing market (FAOSTAT).

Possibly affected technologies and their competitiveness

Pork can be produced in all countries worldwide. Like most industrial productions, pork-production has a clear tendency that larger production units can be run more efficiently than smaller units (economy of scale). Therefore the most competitive production will most likely be an intensive production.

The Danish production of pigs is to some extent limited by public regulation of manure distribution on farmland. However, since this is increasingly overcome by longer transport of the manure, the production can be considered unconstrained.

Danish pig producers have been divided into 9 main categories, and the reaction of the pig producers on a slight increase in market price (i.e. an increased demand) is estimated by modelling in the econometric model ESMERALDA (Jensen et al., 2001). The model is based on market related experiences including effects from public regulation of e.g. manure distribution, and predicts that an increased demand for pork will be met by a varying increases of pig production in pig farm categories (see Table 1). The production, which is affected by a small change in demand, is therefore a weighted average of production in these farms.

According to modelling in ESMERALDA, the pig-producing farm of low intensity, situated on sandy soil would react stronger to a change in pork demand than any other farms (see Table 1). This single farm-type was selected as a reasonable representative for the marginal producers of pigs, because estimates of environmental data for this farm type were more accurate than for other farm types due to a better modelling of the environmentally significant production of grower pigs.

Table 1. The classification of pig-producing farms their contribution to total Danish pig production in 1999, and their predicted contribution to a small increase of production. Dejgaard and Andersen, (2003).

Main-product of the farm	Soil-type	Intensitivity of production (animal density)	Contribution to total Danish pig production (%)	Contribution to affected production (%)
Sugar beet	Clay		3.9	6.5
Seeds	Clay		3.4	4.6
Pigs	Clay	Low	3.7	10.3
Pigs	Clay	Medium	4.4	4.2
Pigs	Clay	High	23.1	3.7
Potatoes	Sand		1.3	6.7
Pigs	Sand	Low	11.1	32.7
Pigs	Sand	Medium	4.7	5.3
Pigs	Sand	High	34.1	10.3
Residual group			10.3	15.7
All farmtypes			100.0	100.0

Processes affected by the demand for cheese

The nature of the market for cheese

Cheese is produced on specialised cheese-dairies with input of milk and output of cheese, whey and cream. Production is increasing in Denmark and globally (FAOSTAT). The market for cheese is strongly influenced by the public regulations (milk quotas) and is therefore not affected directly by a small change in demand at least in a short time perspective.

A screening showed that the cheese production itself had only a small contribution to the environmental impact from cheese. Instead the most important process to identify correctly appeared to be the production of milk, or - to phrase it more precisely - *the processes affected by the use of milk*. Therefore focus turned from the affected cheese producing technology, to the affected milk supplier.

Possibly affected technologies and their competitiveness

A change in the cheese-production does **not** affect milk production in agriculture, because quotas limit the Danish production of milk. Demand for milk for cheese production must therefore influence other processes using milk.

Milk from agriculture can be used to produce drinking milk, cheese, yoghurt, butter and dry milk. Following the dynamics of market economy, the production with the lowest alternative costs (i.e. lowest revenue) will be decreasing its production.

Milk powder is produced in excess and donated to developing countries (Nielsen et al., 2003). Personal communication with the production planners at the dairies indicates that the dry milk production is increased or decreased as demand for other milk products changes (Weidema, 2003). Hence a change in demand for cheese will induce a change in dry milk production and donation. Since dry milk is donated to developing countries it is assumed that it does not displace any other products at these markets and that only dry milk production is influenced by a slight change in demand for cheese. It has not been possible to justify the later assumptions and this point is still open for discussion and development.

Example of the importance of identification of affected processes

The processes that are affected by the demand for cheese as identified above are shown in Figure 1. Figure 2 shows an alternative system delimitation where market regulation of milk production is neglected. This could be relevant for a scenario analysis to assess environmental impacts from cheese when and if the quota system is abolished.

The figures should be read like this: Boxes refer to production processes. Names of grey boxes refer to the main product of the processes. Red arrows represent material or energy transfer between two processes; green arrows represent saved material or energy transfer as a result of displacements and green lines represent displacements. The numbers and the red/green-bars in the boxes show each process' cumulated contribution to total environmental impact measured as a single score indicator. Numbers (Pt) indicate amount of "personal equivalents", which is the environmental impact from Danish production per inhabitant. Please note that the environmental indicator of one process cannot directly be calculated as the sum of the environmental indicators of the processes at the below level. This is due to the looped structure of the diagram, where e.g. the cumulated indicator for electricity in Figure 1 is partly passed on to the cheese-production, partly to the production of whey powder. The processes and figures can be found with further explanation in the SimaPro-database available in Nielsen et al. (2003). Figure 1 shows that the total environmental impact from consumption of 10 tons cheese is equivalent to 1 personal equivalent. This impact is the sum of increased environmental impacts in the cheese-production itself as well as in the processing of by-products (whey and cream). Some reduced environmental impacts are subtracted, because cheese production leads to decreased production of milk powder, and because the whey powder is used as animal feed, and thus replaces barley.

Figure 2 shows that if the quota regime were abolished, the main environmental impact from cheese would lie in the agriculture's production of milk and their inputs. Two productions would be reduced (green bars): the meat from the milking cows would replace alternative production of meat, and the production of rape seed would decreased, because of the milk-farm's by-production of these, and because the soy oil produced as by-product to the soy-meal would replace rape seed oil.

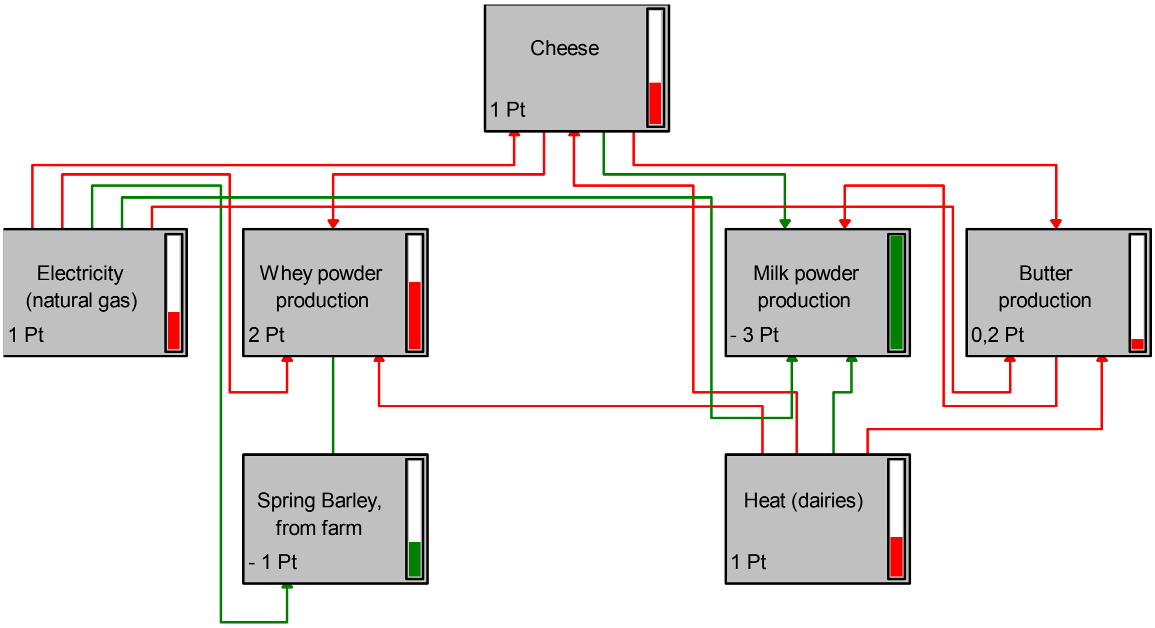


Figure 1. The cumulated environmental impact in the product chain for 10 ton of cheese. All shown processes will be affected by a change in demand for cheese under the present market structure. Only processes contributing with more than 20% of total score are shown.

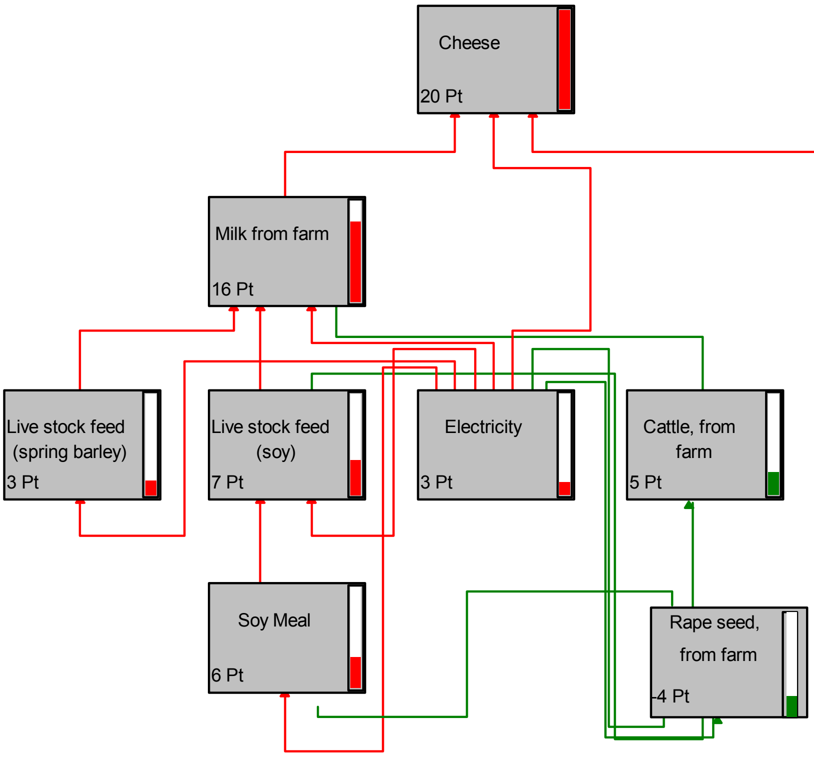


Figure 2. The cumulated environmental impact in the product chain for 10 ton of cheese under the assumption that a milk farm will be affected by an extra production of cheese. Only processes contributing with more than 15% of total score are shown.

It can be seen that the two different identifications of affected technologies yield highly different results. In a comparison with e.g. other products for sandwich filling, the difference between the results in figure 1 and 2 is enough to significantly change the conclusions. If data are used to identify the hotspot-processes in the product chain of cheese, the conclusions will also be irrelevant and misleading.

Discussion and outlook

In the LCAfood database, environmental data for the Danish production of basic food is modelled using system expansion. The affected processes are identified based on quantified as well as more qualitative knowledge on market structures and production economics. This work led us to some key-findings.

Necessary cooperation between production engineers and market-economists

Lifecycle assessment used to be a matter of inputs and outputs from processes linked to each other by a physical flow of materials. However, other processes than those directly involved in the physical chain are influenced by a product demand and these processes influence the final result of an assessment, as shown in the cheese-example above. Hence, lifecycle assessment is no longer just a natural science accomplished by environmental departments and process engineers in co-operation, but a combined natural and social science, which also requires inputs from marketing and economic departments.

The level of details in identification of affected technologies

The level of details in the identification of the affected producer can vary. In the pork-example, the affected producer was identified according to some key factors in the production technology (namely the production intensity, soil type and main production of the farm). In the case of cheese, it was found that the limiting factor to decrease uncertainty of the assessment was not to identify the cheese-producing technology, but to identify which processes was affected by the cheese-production's consumption of milk. This can be seen from the big differences between Figure 1 and 2. Therefore the level of details with which the affected producer is identified should depend upon the factors, which affect the result of the assessment the most.

The workload of market-based LCA

Market-based identification of affected technologies adds a new task to the life cycle assessment. Since this will be on the expense of other tasks of the assessment, it should only be added, if it actually increases the explanatory power of the study. The workload for the identification of system delimitation will increase, especially if the economical information must be found outside the group performing the LCA. Workload for data collection may be increased, because it is less possible to use standard databases, which are based on average data. On the other hand, identification of affected processes prior to the data-collection can help focus the data-collection on the processes, which are most important to the result, and can thus avoid collection of unnecessary data. The advantage of the approach comes in the assessment. Conclusions based on allocation can be very unsatisfactory, if different allocations yield different results. Conclusions based on system-expanded data where affected technologies are identified are easier to interpret, even if there is doubt about the right identification of affected technology, because there is a clear bond between the market-assumption and the different results.

References

FAOSTAT. FAO Statistical Databases. Available at http://apps.fao.org/

Guinée, J. B., Gorrée, M., Heijungs, R., Huppes, G., Kleijn, R., Koning, A. de, Oers, L. van, Sleeswijk, A. W., Suh, S., Udo de Haes, H. A., Bruijn, H. de, Duin, R. van, Huijbregts, M. A. J., Lindeijer, E., Roorda, A. A. H., Ven, B. L. van der, Weidema, B. P. (2001). LCA – An operational guide to the ISO standard. Centre of Environmental Science, Leiden University.

Hardwick, P., Khan, B., Langmead, J. (1990). An introduction to modern economics. 3rd edition. Longman Group UK Limited.

Jensen, J.D., Andersen, M. & Kristensen K. (2001). A Regional Econometric Sector Model for Danish Agriculture – A Documentation of the Regionalized ESMERALDA Model. Danish Research Institute of Food Economics, report no. 129.

Jensen, J.D. and Andersen, M. (2003). Marginal producers of a selection of agricultural products. Working paper. Danish Research Institute of Food Economics. In press.

Kramer, K.J. (2000). Food Matters – On reducing energy use and greenhouse gas emissions from household food consumption. Rijkuniversiteit Groningen

Nielsen P.H., Nielsen A.M., Weidema B.P., Dalgaard R. and Halberg N. (2003). LCA food database. www.lcafood.dk/database.

Weidema, B.P., Frees, N. and Nielsen, A.M. (1999). Marginal Production Technologies for Life Cycle Inventories. *International Journal of LCA* **4** (1), 48 – 56.

Weidema, B. P. (2001). Avoiding Co-Product Allocation in Life-Cycle Assessment. *Journal of Industrial Ecology* **4** (3), 11 - 33.

Weidema B.P. (2003). Market information in life cycle assessments. Environmental Project no. 863. Danish Environmental Protection Agency.

Land occupation and transformation in the Swiss life cycle inventory database ecoinvent 2000

Jungbluth, N., Frischknecht, R.
**ESU-services, Kanzleistrasse 4, CH-8610 Uster, T. +41 1 940 61 91 or +41 1 940 6132, F. +41 1 940 61 94, jungbluth@esu-services.ch, frischknecht@ecoinvent.ch*
Centre for life cycle inventories in the ETH domain, c/o EMPA, CH - 8600 Dübendorf

1. Abstract

Land occupation and land transformation gets more and more attention in life cycle inventory analyses and life cycle impact assessment methods. It is especially important for agricultural and forestry products. However, consistent land occupation and transformation figures for unit processes of an economic sector or even complete process networks including agriculture, forestry, energy supply, transport and waste treatment services, materials production, etc. are still rare. For the Swiss ecoinvent 2000 project, one emphasis is put on a systematic registration and quantification of land occupation and transformation. For that purpose the ecoinvent 2000 project group developed a simplified methodology that allows for a relatively efficient data compilation and data handling while at the same time minimising information loss in view of future developments in LCIA (life cycle impact assessment). The approach uses the CORINE land cover typology which has punctually been enlarged (e.g., land occupation and transformation on the sea bottom, the so-called benthos) to cover all LCI specific land types. It considers as much as possible the experiences and recommendations of the SETAC LCIA working group. Occupation on the one hand and the land types just before and after a land transformation on the other hand are reported separately. Hence, the three types of elementary flows for land use are: "occupation, …" (in m^2.a), "transformation, from …" and "transformation, to …" (both in m^2). On the level of cumulative LCI results, the balance of the total surfaces transformed, for instance "transformation, to forest" minus "transformation, from forest", indicates whether the surface of forests de- or increased (negative and positive balance, respectively) due to the supply of the functional unit at issue. The presentation explains and substantiates the method applied with the help of a case study. The developed methodology can be used to record land use patterns in life cycle inventories for all types of products.

Keywords: land use, occupation, transformation, life cycle inventory

2. Introduction

Land occupation and land transformation gets more and more attention in life cycle inventory analyses and life cycle impact assessment methods (Lindeijer et al. 2001). It is especially important for agricultural and forestry products. However, consistent land occupation and transformation figures for unit processes of an economic sector or even complete process networks including agriculture, forestry, energy supply, transport and waste treatment services, materials production, etc. are still rare and the documentation is not sufficient for an in depth impact assessment. For the Swiss ecoinvent 2000 project, one emphasis is put on a systematic registration and quantification of land occupation and transformation (ecoinvent Centre 2004). For that purpose the ecoinvent 2000 project group developed a simplified methodology that allows for a relatively efficient data compilation and data handling while at the same time minimising information loss in view of future developments in LCIA.

3. Land use and its impacts

The impact category “land use” in LCIA covers a range of consequences of human land use. Land use mainly has impacts on the following area’s of protection (Guinèe et al. 2001):

- Natural Resources
- Natural Environment
- Man-made Environment

According to Lindeijer et al. (2001) the impacts of land use are classified in four groups. Certain land use types might also have a positive impact:

1. Increase of land competition
2. Degradation of biodiversity
3. Degradation of life support functions
4. Degradation of cultural values

The methodology used in the ecoinvent database concentrates on the second and - as far as possible - the third impact (Frischknecht et al. 2004).

4. Distinction between land occupation and land transformation

A distinction is made between

- *land occupation* (i.e., the operation of a power station hinders the occupied land from changing to state it would have under uninfluenced conditions), and
- *land transformation* (i.e., a new assembly plant for airbus airplanes require the conversion of a former natural resort to industrial land; a gravel-pit is converted to a natural resort by active re-cultivation).

For land occupation the surface as well as the duration required for the production of a certain amount of products and services are important. That is why land occupation is recorded in square metres times time (m^2a).

Clearly defined and relatively short changes in the land use type are recorded as land occupation (e.g., the construction of underground natural gas pipelines, which converts agricultural land to an excavation site). For these construction processes as well as for active restoration activities after decommissioning, the land use category "land occupation, construction site" is applied.

Land transformation links a state during an economic activity with a state before and a state after that activity (road construction, power plant erection, active mine restoration, etc.). But it also may occur during the economic activity itself (open pit lignite extraction).

For particular processes the land use type before starting the activity may well be known. However, it is difficult to assess in detail all the land use types which have been converted by the production processes recorded within the ecoinvent project. If not known, the land use type "transformation, from unknown" is applied. Continental or regional statistics about land transformation over time may then be used later on to attribute specific land use type to this land use type "unknown".

Land transformation consists of an entry
1. land transformation, from land use type X, and
2. land transformation, to land use type X.

For land transformation at the beginning of an economic activity the land use type encountered at that point in time is recorded. This starting state, such as "transformation, from forest", is recorded in m^2. The transformation to the land use type valid during the economic activity is recorded as well. For gravel extraction, for instance, the m^2 "transformation, to mineral extraction site" are recorded.

This land transformation needs to be attributed to the total amount of products and services delivered (the life time production of a power plant, one production cycle of a forest, time period until the depletion of a mine, etc.).

Tabel 1 shows the time periods applied in the ecoinvent project if no specific information is available (Frischknecht et al. 2004).

Active restoration at the end of an extraction or production process is modelled as a separate unit process (restoration, gravel-pit", "restoration, copper mine"). This process includes technical requirements such as diesel for construction machines, seeds, etc.. Additionally, land transformation to the final land use type (and from the land use type during the operation of the economic process) are recorded (e.g., "transformation, from mineral extraction site", and "transformation, to pasture and meadow"). Such restoration processes may be required by an infrastructure process (such as a power plant: "restoration, power plant") or by the production process itself (in case land transformation and restoration takes place during the economic activity (e.g., lignite extraction).

Land transformation caused by the use of the land for new purposes is attributed to this future new uses. No land transformation after the assumed life time is recorded for actual land uses that are likely not change in the future (such as transport infrastructure, agricultural land) as well as for land abandoned and subjected to natural succession.

5. Regional differentiation

Inventory data are collected on the level of national averages. Hence, no regional differentiation can be made. Unit processes are described by a geographic code, be it a country or a continent or an international organisation. This geographic code provides information about where the land occupation and transformation of the process at issue takes place. However, the ecoinvent database does not yet allow for an automatic evaluation of this information.

Table 1. Land occupation types used in ecoinvent based on the CORINE land cover types classification. The same types are used for land transformation, according to the naming rules as described in the text.

english name	CORINE class	Use period
Occupation, arable	CORINE 21	1
Occupation, arable, non-irrigated	CORINE 211	1
Occupation, arable, non-irrigated, diverse-intensive	CORINE 211b	1
Occupation, arable, non-irrigated, fallow	CORINE 211c	1
Occupation, arable, non-irrigated, monotone-intensive	CORINE 211a	1
Occupation, construction site	CORINE 133	not considered
Occupation, dump site	CORINE 132	10
Occupation, dump site, benthos	CORINE 132a	1
Occupation, forest	CORINE 31	80
Occupation, forest, extensive	CORINE 31a	100
Occupation, forest, intensive	CORINE 31b	80
Occupation, forest, intensive, clear-cutting	CORINE 31b2	80
Occupation, forest, intensive, normal	CORINE 31b1	60
Occupation, forest, intensive, short-cycle	CORINE 31b3	30
Occupation, heterogeneous, agricultural	CORINE 243a	100
Occupation, industrial area	CORINE 121	50
Occupation, industrial area, benthos	CORINE 121c	??
Occupation, industrial area, built up	CORINE 121a	50
Occupation, industrial area, vegetation	CORINE 121b	50
Occupation, mineral extraction site	CORINE 131	20
Occupation, pasture and meadow	CORINE 231	30
Occupation, pasture and meadow, extensive	CORINE 231b	20
Occupation, pasture and meadow, intensive	CORINE 231a	20
Occupation, permanent crop	CORINE 22	20
Occupation, permanent crop, fruit	CORINE 222a	15
Occupation, permanent crop, fruit, extensive	CORINE 222b	15
Occupation, permanent crop, fruit, intensive	CORINE 222a	15
Occupation, permanent crop, vine	CORINE 221	25
Occupation, permanent crop, vine, extensive	CORINE 221b	25
Occupation, permanent crop, vine, intensive	CORINE 221a	25
Occupation, sea and ocean	CORINE 523	??
Occupation, shrub land, sclerophyllous	CORINE 323	100
Occupation, traffic area, rail embankment	CORINE 122d	50
Occupation, traffic area, rail network	CORINE 122c	100
Occupation, traffic area, road embankment	CORINE 122b	50
Occupation, traffic area, road network	CORINE 122a	100
Occupation, unknown	CORINE x	1
Occupation, urban, continuously built	CORINE 111	80
Occupation, urban, discontinuously built	CORINE 112	80
Occupation, water bodies, artificial	CORINE 512a	100
Occupation, water courses, artificial	CORINE 511a	100

6. Naming rules

The differentiation between transformation and occupation is reflected in the naming of land use elementary flows. It takes pattern from the naming proposals of a Dutch project (Lindeijer & Alfers 2001) and deviates from the provisional proposals of the SETAC Europe working group (de Beaufort-Langeveld et al. 2003; Hischier et al. 2001):

- Occupation, *type, subtype*
- Transformation, from *type of occupation*
- Transformation, to *type of occupation*

The different levels of details in describing the land use type are separated by commas:

- Occupation, arable
- Occupation, arable, non-irrigated
- Occupation, arable, non-irrigated, monotone-intensive

The highest level of information detail is always used and recorded in the inventories.

7. Land use types in ecoinvent

The definition of land use types takes pattern from the CORINE land cover types (Bossard et al. 2000). New, and partly more detailed land use types have been added to the CORINE table using the same systematic. Only land cover types required in the ecoinvent project have been added. The list may however easily be extended if required.

Land use types do not include national or even regional differentiation. For instance, the land use type "pasture and meadow, extensive" covers land occupation (and transformation) by Alpine pastures as well as South American cattle pastures.

If the land use types before and after the operation phase of the economic activity are not known, land transformation is recorded with "transformation, from unknown" and "transformation, to unknown", respectively. With that, the sum of all "transformation, to ..." equals the sum of all "transformation, from ...". From these figures the net transformation of land use types can be calculated.

If the total amount of m^2 "transformation, from forest" is larger than the amount of m^2 "transformation, to forest", calculated for the production of 1kWh of Swiss low voltage electricity, the production of this kWh reduced the total amount of forest.

8. Allocation issues

Land transformation and occupation may be allocated among several products or services. Roads for instance are built for personal and freight traffic. Allocation is done with regard to the influence of each of the products / services on the impact categories of land occupation and transformation listed above.

9. Attribution of land use to operation or construction/dismantling (infrastructure)

Land occupation and transformation may in some cases either be attributed to the infrastructure or the operation of a process. The surface of a greenhouse may rather be recorded in the infrastructure part whereas the farm land would rather be recorded in the operation of an agricultural process. As a rule, agricultural and forest property surfaces are attributed to the operation as long as they do not include buildings and roads. Land use by buildings, forest roads, greenhouses and the like are attributed to the infrastructure.

Ecoinvent allows for calculation of results excluding infrastructure requirements. Hence, land used by infrastructure is neglected and may influence impact assessment results substantially. Care must be taken when comparing LCIA results computed without infrastructure contributions.

10. Example

The ecoinvent approach to land use is illustrated with the following example of gravel extraction:

- total area (1000 m*1000 m = 1'000'000 m^2),
- gravel-pit used during 20 years,
- 1'000'000 tons of gravel extracted per year,
- 2 years of restoration activities,
- diesel consumption of 500 t (21.3 TJ) per year during extraction and of 100 t (4.3 TJ) during restoration.

The inventories for the two processes "gravel, at extraction/kg/RER/0" and "restoration, gravel-pit/m^2/RER/0" are shown in Tabel 2. Land occupation is calculated by dividing the total surface by the total amount of gravel extracted per year. Land transformation is calculated by dividing the total surface by the total life time production (1'000'000m^2 / 20a * 1'000'000'000 kg/a = $5.0 * 10^{-5}$ m^2/kg). The requirements for restoration are also evenly attributed to the life time production of the gravel-pit. Restoration leads to a forest (assumption). This restoration activity may not be included in the inventory of forestry products (e.g., timber). For timber production, the correct land transformation before starting that activity is "transformation, from forest". In the column results one can see that the total surface transformed is $5.0 * 10^{-5}m^2$ per kg gravel extracted and that the net transformation is "from unknown" to "forest". The two transformations "to mineral extraction site" and "from mineral extraction site" cancel each other out.

Table 2. Example of unit process raw data including land transformation and land occupation.

			raw data		results
			gravel, at extraction	restoration, gravel-pit	gravel, at extraction
			kg	m^2	kg
Resource use	Occupation, mineral extraction site	m^2a	0.001		0.001
	Occupation, construction site	m^2a		2	0.0001
	Transformation, from unknown	m^2	5.00E-05		5.00E-05
	Transformation, to mineral extraction site	m^2	5.00E-05		5.00E-05
	Transformation, from mineral extraction site	m^2		1	5.00E-05
	Transformation, to forest	m^2		1	5.00E-05
Technosphere Inputs	restoration, gravel-pit	m^2	5.00E-05		5.00E-05
	diesel, burned in building machine	MJ	0.021	4.3	0.0212
	...	...			
Output	gravel, at extraction	kg	1		1
	restoration, gravel-pit	m^2		1	

11. Conclusions

The methodology developed within the ecoinvent 2000 project can be used to record land use patterns in life cycle inventories for all types of products. It fits for the implementation of different LCIA methodologies. The experiences within the ecoinvent project showed that the method is quite easy to apply for the recording of land occupation. But, it is still time consuming to investigate the land transformation patterns for a given process because the necessary data are quite often not available. Therefore it might be useful to elaborate clear guidelines how this transformation can be investigated.

12. References

Bossard M., Feranec J. and Otahel J. (2000) CORINE land cover technical guide - Addendum 2000. 40. European Environment Agency (EEA), Copenhagen (DK), retrieved from: www.eea.eu.int, http://etc.satellus.se/I&CLC2000/download.htm.

de Beaufort-Langeveld A. S. H., Bretz R., van Hoof G., Hischier R., Jean P., Tanner T. and Huijbregts M. (2003) Code of Life-Cycle Inventory Practice (includes CD-ROM). SETAC, retrieved from: www.setac.org.

ecoinvent Centre 2004. ecoinvent Centre (2004) ecoinvent data v1.1. ISBN 3-905594-38-2. Swiss Centre for Life Cycle Inventories, Dübendorf, CH, retrieved from: www.ecoinvent.ch.

Frischknecht R., Jungbluth N., Althaus H.-J., Doka G., Dones R., Hischier R., Hellweg S., Nemecek T., Rebitzer G. and Spielmann M. (2004) Overview and Methodology. Final report ecoinvent 2000 No. 1. Swiss Centre for Life Cycle Inventories, Dübendorf, CH, retrieved from: www.ecoinvent.ch.

Guinèe J. B., Gorrée M., Heijungs R., Huppes G., Kleijn R., de Koning A., van Oers L., Wegener Sleeswijk A., Suh S., Udo de Haes H. A., de Bruijn H., van Duin R., Huijbregts M. A. J., Lindeijer E., Roorda A. A. H. and Weidema B. P. (2001) Life cycle assessment; An operational guide to the ISO standards; Parts 1 and 2. Ministry of Housing, Spatial Planning and Environment (VROM) and Centre of Environmental Science (CML), Den Haag and Leiden, The Netherlands.

Hischier R., Baitz M., Bretz R., Frischknecht R., Jungbluth N., Marheineke T., McKeown P., Oele M., Osset P., Renner I., Skone T., Wessman H. and de Beaufort A. S. H. (2001) Guidelines for Consistent Reporting of Exchanges from/to Nature within Life Cycle Inventories (LCI). *In: Int. J. LCA*, **6**(4), pp. 192-198.

Lindeijer E., Müller-Wenk R., Steen B., with written contributions from, Baitz M., Broers J., Finnveden G., ten Houten M., Köllner T., May J., Mila i Canals L., Renner I. and Weidema B. (2001) Impact Assessment of resources and land use. SETAC WIA-2 taskforce on resources and land.

Lindeijer E. and Alfers A. (2001) Summary of Step A of the Delfts Cluster Research Programme on Land Use in LCA. *In: Int. J. LCA*, **6**(3), pp. 186.

A new land use impact assessment method for LCA: theoretical fundaments and field validation

Peters, J. (1), García Quijano, J. (2), Content, T. (3), Van Wyk, G. (4), Holden, N.M. (1), Ward, S.M. (1) and Muys, B (3,).*

(1) Department of Agricultural and Food Engineering, University College Dublin, Ireland
(2) Division of Engineering and Applied Sciences, Harvard University, USA
(3) Laboratory for Forest, Nature and Landscape Research, Katholieke Universiteit Leuven, Vital Decosterstraat 102, B-3000 Leuven, Belgium, Tel: +32 16329726, Fax: +32 16329760, bart.muys@agr.kuleuven.ac.be
(4) Faculty of Agricultural and Forestry Sciences, University of Stellenbosch, South Africa

Abstract

For LCA studies with a major part of the life cycle in agriculture and forestry, incorporation of a land use impact category is essential. Different method proposals have been formulated in the last 6-7 years, but their common weakness is an arbitrary choice of indicators due to a lack of solid theoretical basis. Other frequently reported problems of the existing proposals are related to the choice of a reference system, the lack of universal applicability and time/space issues such as the incompatibility between land use change and permanent land occupation. The new method proposal addresses most of these problems. A theoretical background based on ecosystem thermodynamics uses the hypothesis that in absence of human land use impact, all ecosystems tend to maximize the internal exergy level and control over incoming and outgoing exergy fluxes. In order to measure land use impact, the deviation from the site specific maximum ecosystem performance in exergy terms is estimated using 17 quantitative indicators and aggregated into four thematic scores. The indicator scores for ecosystem biomass and structure and for biodiversity are quantifying the land use impact on the ecosystem exergy level, while the indicator scores for water and for soil and nutrients are quantifying the land use impact on ecosystem buffering capacity for exergy fluxes. Thematic scores are multiplied by the area x time needed for the production of the functional unit. Test results from different land uses in several countries under cold temperate, mediterranean, subtropical and tropical climates indicate that the method is workable and universally applicable. Results are discussed and recommendations for further improvements are formulated.

Keywords: Life Cycle Assessment, land use impact assessment

1. Introduction

Many human activities have important spatial needs (e.g. for the extraction of resources, for production processes, for landfill). The use of land for a given production process will often make the land unavailable for other uses. Land transformation and occupation may also change the “quality” of the land. The land use impact category is especially relevant in LCA studies of products with a major part of their life cycle in the agriculture and forestry sector. *Several methods have been proposed to assess the land use impact (Sturm and Westphal,* 1996, Baitz et al., 1998, Bradley et al., 1998, Giegrich and Sturm, 1998, Lindeijer et al., 1998 Schweinle, 1999, Køllner, 1999). The strengths and weaknesses of these proposals and the differences between them have been tested in some comparative studies (Giegrich et al., 1999, Leplae, 2000, Peters, 2002). Two expert groups published recommendations for developing better land use impact assessment methods: the COST E9 action (LCA for forestry and forest products) working group on land use (Schweinle et al., 2002) and the SETAC working group on resources and land use (Lindeijer, 2002).

In this paper a newly developed method based on these recommendations is presented. It starts from a scientific concept for the choice of indicators, as recommended by COST E9 (Schweinle et al., 2002). For the impact assessment it combines the strong points from the methods of Sturm and Westphal (1996), Baitz et al. (1998), Giegrich and Sturm (1998) Lindeijer et al. (1998), and Schweinle (1999), and adds some new ideas, while remaining simple and flexible.

2. Methods

2.1. Basic concept and reference system

COST E9 working group 2 proposed to use ecosystem exergy as a basic concept (Schweinle et al., 2002). Ecosystems are systems that are open to energy and/or material flows and reside in states of thermodynamic non-equilibrium. When in thermodynamic non-equilibrium, the second law of thermodynamics cause ecosystems to counter applied gradients and oppose movement towards equilibrium (Schneider and Kay, 1994). Ecosystems attempt to develop away from the thermodynamic equilibrium by gaining exergy and losing entropy. The goal function of an ecosystem can be defined as the continuous tendency to go as far as possible from the thermodynamic equilibrium by: (i) exergy storage maximisation (Bendoricchio and Jørgensen, 1997, Fath et al., 2001, Scott, 2003) and (ii) maximising external exergy flow dissipation (Schneider and Kay, 1994, Fath et al., 2001). Maximisation of exergy storage, makes ecosystems build up biomass and embedded information (genetic heritage) (Bendoricchio and Jørgensen, 1997) in complex structures with greater biodiversity and more hierarchical levels. The ecosystems tendency to maximize exergy dissipation results in more exergy capture and flow in the system, more energy and nutrient cycling within the ecosystem, higher trophic structures, more respiration and transpiration, more ecosystem biomass and a greater organism diversity (Schneider and Kay, 1994). It is hypothesised that for any site (i.e. a combination of abiotic circumstances such as climate, soil, aspect) the potential natural vegetation (i.e. the climax system) is the ecosystem with greatest possible exergy storage and dissipation level under natural circumstances (Muys et al., 2001). The potential natural vegetation is chosen as the reference system in this method. Any human activity altering physical, chemical or biological components of an ecosystem will have a direct impact on the exergy storage and dissipation (Schweinle et al., 2002). Measuring exergy storage and dissipation could provide a measure of land use impact on ecosystems. Direct exergy storage calculation has not been possible (Scott, 2003), but indirect top-down approaches using indicators have been proposed (Luvall et al., 2001, Schweinle et al., 2002).

The exergy concept may be linked with environmental issues of concern which are grouped in three areas of protection (or safeguard) (International Organisation for Standardisation, 1997, Udo de Haes, 1999): (i) natural environment (e.g. human interventions causing change in exergy storage or dissipation results in ecosystem transition from climax to a secondary state);

(ii) natural resources (e.g. desertification reduces exergy storage and dissipation and adversely impacts on the economic value of land) and (iii) human health (e.g. biomass/biodiversity decline reducing supply of plant materials or changes to soil processes causing health problems (e.g. Oliver, 1997)).

2.2. Land use impact calculation

The considered land must be classified as site types with homogenous abiotic variables. For every site type, the reference system, which is the potential natural vegetation, must be identified. The method describes the land use impact by 17 quantitative impact indicators divided over 4 themes: soil, water, vegetation and biodiversity (Table 1). A score (ΔQ) is attributed to the chosen indicators that reflects the difference in quality between the actual situation and the reference. In exergy terms, human land use can affect the ecosystem quality in two ways: it can directly change the exergy storage level and the ability of the ecosystem to control exergy flows (e.g. removal of living biomass), and it can change the site quality (e.g. by irrigation, fertilisation, etc.). Changing the site will lead to another climax system with another maximum exergy level. The indicators are chosen in such a way that the reference has a value equal to or close to zero and that impacts cause a positive value with a maximum of 100. A natural ecosystem will get an impact score (close to) zero, while anthropogenically modified ecosystems (with lower exergy levels) will get a positive impact score proportional with the impact magnitude. It is however possible that human interventions (e.g. introduction of new species) could increase the buffering capacity of the ecosystem without changing the site quality. Therefore, a negative impact score up to –25 is provided for in the method.

Table 1. Impact indicators grouped per theme: soil, water, vegetation and biodiversity.

Code	Indicator	Formula	Units
S1	Soil compaction	$\left(\dfrac{area_{aff} * \left(perm_{ref} - perm_{act}\right)}{area_{tot} * perm_{ref}}\right) * 100$ where $area_{aff}$ = area affected; $area_{tot}$ = total area; $perm_{ref}$ = permeability at the reference state; $perm_{act}$ = permeability at the actual state	$\left(\dfrac{ha}{ha} \times \dfrac{cm/d}{cm/d}\right)$
S2	Soil structure disturbance by ploughing, etc.	$\left(\dfrac{area_{aff} * depth}{area_{tot}} * \dfrac{times_{S2}}{rot}\right) * 100$ where $times_{S2}$ = number of soil works per rotation period; rot = length of rotation period (in years)	$\left(\dfrac{ha * m}{ha} \times \dfrac{yr}{yr}\right)$
S3	Soil erosion	$\left(\dfrac{100 * USLE}{Soil\ depth}\right) * 100$ where USLE = soil loss in t ha^{-1} yr^{-1}; soil depth = total rootable soil depth in t ha^{-1}	$\left(\dfrac{t/ha \cdot yr}{t/ha}\right)$
S4	Cation Exchange Capacity (CEC)	$\left(1 - \dfrac{CEC_{act}}{CEC_{ref}}\right) * 100$	$\left(\dfrac{Meq/100\,g}{Meq/100\,g}\right)$
S5	Base Saturation (BS)	$\left(1 - \dfrac{BS_{act}}{BS_{ref}}\right) * 100$	$\left(\dfrac{\%}{\%}\right)$

W1	Evapotrans-piration (ET)	$\left(1-\frac{ET_{act}}{ET_{ref}}\right)*100$	$\left(\frac{mm/yr}{mm/yr}\right)$
W2	Surface runoff (SR)	$\frac{SR}{P-ET}*100$	$\left(\frac{mm/yr}{mm/yr}\right)$
V1	Total above-ground living bio-mass (TAB)	$\left(1-\frac{TAB_{act}}{TAB_{ref}}\right)*100$	$\left(\frac{t/ha}{t/ha}\right)$
V2	Leaf area index (LAI)	$\left(1-\frac{LAI_{act}}{LAI_{ref}}\right)*100$	$\left(\frac{m^2/m^2}{m^2/m^2}\right)$
V3	Vegetation height (H)	$\left(1-\frac{H_{act}}{H_{ref}}\right)*100$	$\left(\frac{m}{m}\right)$
V4	Free Net Primary Production (fNPP)	$\left[1-\left(\frac{NPP_{act}-AH}{NPP_{ref}}\right)\right]*100$ where AH = annual harvest	$\left(\frac{t/ha\,yr - t/ha\,yr}{t/ha\,yr}\right)$
V5	Crop biomass	$\left(\frac{crop\,biomass}{total\,biomass}\right)*100$	$\left(\frac{t/ha\cdot yr}{t/ha\cdot yr}\right)$
B1	Artificial change of water balance	$\left[\left(\frac{area_{irr}+area_{drain}}{total\,area}\right)\right]*100$ where $area_{irr}$ = irrigated area; $area_{drain}$ = drained area	$\left(\frac{ha}{ha}\right)$
B2	Liming, fertilisa-tion, empoverish-ment	$\left[\left(\frac{area_{aff}}{area_{tot}}*\frac{times_{B2}}{rot}\right)\right]*100$ where $times_{B2}$ = number of applications per rotation period	$\left(\frac{ha}{ha}\times\frac{yr}{yr}\right)$
B3	Biocides	$\left[\left(\frac{area_{aff}}{area_{tot}}*\frac{times_{B3}}{rot}\right)\right]*100$ where $times_{B3}$ = number of applications per rotation period	$\left(\frac{ha}{ha}\times\frac{yr}{yr}\right)$
B4	Canopy cover of exotic plant spe-cies (Ex)	$\left(\frac{Ex}{total\ cover}\right)*100$	$\left(\frac{number}{number}\right)$
B5	Number of plant species (Sp)	$\left(1-\frac{Sp_{act}}{Sp_{ref}}\right)*100$	$\left(\frac{number}{number}\right)$

The 17 different indicator scores are aggregated in 4 thematic scores for soil (ΔQ_S), water (ΔQ_W), vegetation (ΔQ_V) and biodiversity (ΔQ_B). The aggregation is done as follows:

$$\Delta Q_S = \frac{\sum_{i=1}^{n}\Delta Q_{Si}}{N} \quad where \quad N = 5 \qquad \text{Eq.(1).}$$

$$\Delta Q_W = \frac{\sum_{j=1}^{m}\Delta Q_{Wj}}{M} \quad where \quad M = 2 \qquad \text{Eq.(2).}$$

$$\Delta Q_V = \frac{\sum_{p=1}^{x} \Delta Q_{Vp}}{X} \quad where \quad X = 5 \qquad \text{Eq.(3).}$$

$$\Delta Q_B = \frac{\sum_{q=1}^{y} \Delta Q_{Bq}}{Y} \quad where \quad Y = 5 \qquad \text{Eq.(4).}$$

The soil and water themes reflect the buffering capacity of the ecosystem for exergy flows. The vegetation and biodiversity themes reflect the exergy level of the ecosystem itself, in terms of biomass, complex trophical networks and genetic information.

Land use denotes the use of a piece of land for a certain purpose during a certain period of time. So space and time are essential parameters. The dimension of land use is therefore area x time per functional unit (FU), i.e. the area needed to produce 1 functional unit in 1 rotation period. The final thematic impact scores are:

$$S_S = \Delta Q_S * (area \times time)_{FU} * FU^{-1} \qquad \text{Eq.(5).}$$

$$S_W = \Delta Q_W * (area \times time)_{FU} * FU^{-1} \qquad \text{Eq.(6).}$$

$$S_V = \Delta Q_V * (area \times time)_{FU} * FU^{-1} \qquad \text{Eq.(7).}$$

$$S_B = \Delta Q_B * (area \times time)_{FU} * FU^{-1} \qquad \text{Eq.(8).}$$

Eq.(5-8) can be applied to cases of land use occupation and land use change. This distinction is proposed by the COST E 9 working group on land use (Schweinle et al., 2002): land use occupation is the continuous use of a certain piece of land for a certain period of time, land use change denotes a more or less abrupt change from one land use to another. A poplar plantation with a rotation period of 25 years is considered as an example of land use occupation. To calculate the mean land use impact over 100 years for the different themes Eq.(9) has to be used. The factor (area x time)$_{FU}$ is assumed to be constant over the rotation periods.

$$S_i = \frac{\sum_{i=0}^{n} \Delta Q_n}{N} * (area \times time)_{FU} * FU^{-1} \qquad \textit{Eq.(9).}$$

for i = soil, water, vegetation and biodiversity

As can be seen in Eq.(10) land use change compares the difference in quality of the last rotation of the former land use with the first rotation of the new land use. A degradation of land quality will lead to positive values, land quality improvement to negative values. The (area x time)$_{FU}$ factor is for the new land use. A visual representation is given in Figure 1.

$$S_i = [\Delta Q_2 - \Delta Q_1] * (area \times time)_{FU} * FU^{-1} \qquad \textit{Eq.(10).}$$

for i = soil, water, vegetation and biodiversity

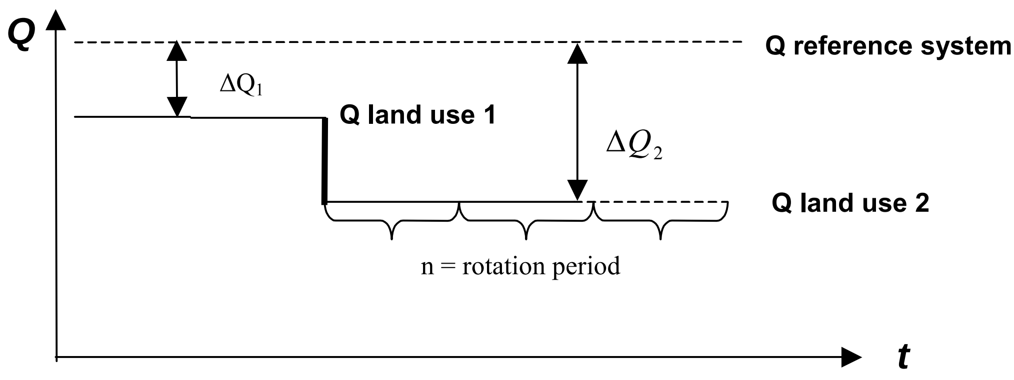


Figure 1. Land use quality (Q) as a function of time (t) during land use change.

2.3. Test scenario definition

The proposed land use impact assessment method has to fit into the LCA framework. Important criteria are that should be met include: universal applicability; the reference system must be definable; the assessment must be quantitative;, a meaningful functional unit must be used; the dimension must be appropriate; linearity; sufficient sensitivity to land use change; and indicator efficiency. To test the method against these criteria different test scenarios were assessed in different climatic and geographical zones.

Multifunctional forest in Flanders (TEMP multi for)
A forest where wood production is combined with high ecological and recreational value, characterized by long rotations (150 years), managed with a thinning frequency of once every 10 years and regenerated with a group selection system. For the simulation we used inventory data of Meerdaal Forest near Leuven, a 1200 ha FSC certified ancient woodland. (Muys et al., 2002).

Short rotation coppice in Flanders (TEMP en for)
Afforested agricultural lands with willow and poplar clones. Fully mechanised harvesting occurs in 3-year rotation cycles for the above-ground biomass and 25-year rotation cycles for the below-ground biomass (Muys et al., 2002).

Eucalypt plantation in Spain (MED eucalypt)
Different Eucalyptus globulus plantations are analysed, mainly in Galicia and Huelva. Rotation periods of 12 years are used to produce pulp (Madueño, 2003).

Tropical rainforest conservation (TROP land use)
Tropical moist forest in South-western Cameroon that is part of the Tropenbos Cameroon Programme. Primary forest, selective logging, shifting cultivation and agricultural land uses are assessed. Logging activities focus on Azobé (Lophira alata Banks ex Gaertn.), Tali (Erythrophleum ivorense A. Chev) and Padouk (Pterocarpus soyauxii Taub). The shifting cultivation system involves the clearing and burning of primary and old secondary forest just be-

fore the rainy season. Cleared land is cultivated and left after a maximum of five years. Agricultural land use consists of cassava cultivation. (Leplea, 2000).

Subtropical agroforestry plantation (SUBTROP land use)
The plantation combines avocado (*Persea americana* Mills) cultivation with Eucalypt (*Eucalyptus grandis* Hill ex Maiden) forestry. A clearcut system is used and leftovers are burned. Eucalypts have rotation cycles of 25 years and are thinned twice and pruned three times. The rotation cycle for avocado varies between 25 and 30 years. Avocado trees are pruned to keep a constant height of 7.5 m. (Content, 2003).

Mediterranean pine plantation (MED land use)
Afforestation of semi -natural scrubland with *Pinus radiata* based on data from the FSC certified Jonkershoek plantation near Stellenbosch, Western Cape, South Africa. The rotation length is 30 years and includes two thinnings and a final clearcut. (Peters, 2002). To test the suitability of the functional unit, the linearity of the method and the dimensions of the end result, results for the pine plantation are expressed per functional unit Eq.(5-8), where one tonne of wood produced is the functional unit. The methods output (thematic scores) for the test scenarios were analysed by Monte Carlo – Chi Square analysis. It is assumed that independent and dependent variables are discrete. The Chi Square test statistic is:

$$\chi^2 = \sum_j \sum_k \frac{\left(O_{jk} - E_{jk}\right)^2}{E_{jk}} \qquad \textit{Eq.(11).}$$

with O_{jk} = Observed value row j, column k;
E_{jk} = Expected value row j, column k

The Monte Carlo test was used because it can handle small expected values, small sample sizes and degrees of freedom do not have to be verified. Monte Carlo also allows for null hypothesis (H_0) variation, depending on the randomisation. H_0 states that distribution is uniform over the whole matrix, rows and columns when respectively the whole matrix, columns and rows are randomised.

2.4. Principal Component Analysis (PCA)

For testing on indicator efficiency, PCA was used. A data matrix of land uses versus indicator scores was constructed, and from the data matrix, a correlation matrix was calculated. Principle Components – orthogonal axes explaining maximum data variation - were defined and presented graphically, with data presented as vectors. Distance between data is proportional to data similarity; angles between vectors are proportional to correlation. Orthogonal projection of vectors on the principal axis are eigenvalues and are proportional to correlation between vector and principle axis.

3. Results

3.1. Land use impact assessment of different test scenarios

All scenarios were divided into homogenous site classes, and for each site class the reference vegetation was identified. Indicator scores were determined and aggregated into thematic scores according Eq.(1-4) and summarised in Table 2a. Thematic scores of different land uses were tested one by one using Monte Carlo – Chi Square analysis with H_0: there is no significant difference between the two land uses analysed. Some important results are:

- Land use impact decreases with decreasing land use intensity. Infrastructure has the greatest impact followed by agricultural land use and forestry. Utilisation of natural resources (TROP primary forest, MED fynbos, MED indigenous forest and SUBTROP kloof forest) had the lowest impact.
- Impacts of intensive land uses (TROP infrastructure, MED infrastructure, SUBTROP agricultural land use) do not differ significantly.
- Reference vegetations suggest that there is no significant difference ($p > 0.05$) between TROP primary forest, MED indigenous forest, and SUBTROP kloof forest, but there are significant difference ($p < 0.05$) between MED fynbos and TROP primary forest, MED indigenous forest and SUBTROP kloof forest.
- There is no significant difference ($p > 0.05$) between the two Eucalypt plantations assessed (MED eucalypt and SUBTROP eucalypt).
- In Flanders, multifunctional forestry (TEMP multi for) has lower impacts on vegetation and biodiversity than energy forestry (TEMP en for). Energy forestry has lower impacts on soil and water. There is no overall significant difference ($p > 0.05$).
- There is no significant difference ($p > 0.05$) between selective logging and shifting cultivation in tropical rainforest (TROP). Though we can conclude that shifting cultivation has a slightly higher impact, because impacts are greater at 0.05 significance level compared with primary forest, and no significance difference ($p > 0.05$) can be seen between primary forest and selective logging.
- Fynbos (MED fynbos) and indigenous forest (MED indigenous forest) are both reference vegetations in the Western Cape. As a consequence impacts are close to zero. Fynbos has a higher water impact than indigenous forest becasue natural burning cycles lead to poor water flux buffering. There is a significant difference ($p < 0.05$) between both land uses.
- Mediterranean pine plantation (MED pine plantation) has a fairly low impact on all themes except for biodiversity. Impact on water is negative, indicating a higher water flux buffering capacity than MED fynbos and MED indigenous forest.
- MED Fire belt has a high impact on vegetation, because it is burned every 4 years, keeping vegetation shorter and less structured than the reference vegetation.
- In the subtropical agroforestry scenario, Avocado has significantly higher impacts than Eucalypt ($p < 0.05$).

- There is no significant difference ($p > 0.05$) between SUBTROP secondary forest and SUBTROP kloof forest impacts, except for vegetation. Secondary forest is not as structured as the reference kloof forest vegetation.

Thematic scores for the different land uses in the Mediterranean pine plantation scenario were scaled spatially to get thematic scores for the whole management unit. Results of Eq.(1-4). are 0.91, 0.47, 6.87 and 10.00 respectively. The (area x time)$_{FU}$ * FU^{-1} is 1,675 m^2 yr t^{-1}, i.e. 1,675 m^2 of pine plantation are needed to harvest 1 t wood after 1 rotation period. Using Eq.(5-8) results in S_S 1524 m^2 yr t^{-1}, S_W 782 m^2 yr t^{-1}, S_V 11507 m^2 yr t^{-1} and S_B 16750 m^2 yr t^{-1} (Table 2b).

3.2. Principal Component Analysis

The complexity of natural systems, and our limited understanding of cause-effect relationships within them means that it is not possible to include all relevant energy and material flows in the impact assessment. It is possible however, to characterise a limited number of impacts using indicators. The method defines 17 quantitative indicators within 4 themes. Indicator choice is based on the exergy concept. To test the indicator efficiency a PCA was used. For all indicators and land uses of the mediterranean pine plantation and subtropical agroforestry plantation scenario a correlation matrix was made and principle components were calculated. Results are presented graphically in Figure 2, and both axes explain 73% of the variation. Distance between points is proportional to similarity, angles between vectors are proportional to correlation. 3 clusters can be distinguished. Soil erosion (S3) is not correlated with other indicators, probably because of the low erosion value for all land uses. Soil structure disturbance (S2), liming and fertilisation (B2) and biocides (B3) are strongly correlated because silvicultural practices always have a related soil disturbance. Furthermore these indicators have a very similar build-up through time. A second cluster consisted of indicators S4, S5, W1, V2 and V3. All these indicators have a similar build –up through time, but correlation also results because of ecological variables (which are difficult to clearly define). It is not surprising that V3 and W1 are correlated, LAI is often used to model evapotranspiration. The third cluster consists of S1, W2, V1, V4, V5, B1, B4, and B5. A possible explanation of correlations between these indicators for the land uses assessed in PCA is that a higher crop percentage per land use results in greater annual harvest (V4), greater crop biomass (V5), greater cover of exotic plants (B4), lower number of species (B5). Soil compaction (S1) and surface runoff (W2) are positively correlated in general.

Table 2. (a) Thematic impact scores for the different land uses; (b) Thematic impact scores per FU for the Mediterranean pine plantation scenario

(a) THEMATIC IMPACT SCORES [/]:		**ΔQ_S**	**ΔQ_W**	**ΔQ_V**	**ΔQ_B**
SCENARIO					
CLIMATIC ZONE	LAND USE				
TEMP	multi for	2.6	5.3	25.1	10.3
TEMP	en for	-1	-3.8	44.5	35
MED	eucalypt	5.8	4	34.4	38.8
TROP	primary forest	0	1.5	0	0
	selective logging	1.4	2.6	9.2	2.4
	shifting cultivation	1.8	4	13.1	17
	agricultural land use	20.5	21	81	52
SUBTROP	eucalyptus	8.2	-4.4	33.3	39.5
	avocado	9.7	24.9	38.7	87.5
	wetland	0.2	-8.9	46.6	17.3
	secondary forest	0	8.25	25.4	1.3
	infrastructure	60	79.2	100	60
	kloofbos	0	7.3	0	0
MED	pine plantation	0.5	-3.7	4.6	15.9
	fynbos	0.5	5	0	0
	indigenous forest	0	0	0	0
	infrastructure	20	47.3	100	40
	fire belt	0.5	5	39.1	0
(b) THEMATIC IMPACT SCORES PER FU [m² yr t^{-1}]:		**S_S**	**S_W**	**S_V**	**S_B**
FU = 1 ton of wood leaving the plantation					
subtropical plantation scenario		1524	782	11507	16750

4. Discussion and Conclusion

The method allowed for comparison of different land uses in different climatic or geographical regions in a fully quantitative matter. The potential natural vegetation varied over all scenarios assessed and was a useful reference. Land use impact can be expressed per FU and the dimensions [m^2 yr FU^{-1}] are suitable for LCA application. The linearity requirement is satisfied, doubling the FU results in a doubled impact score. The methods sensitivity is moderate. All indicators and the methodology are fully quantitative, consequently every change in input results in a changes of one or more thematic impact scores. A weakness is the subjective minimum threshold of –25% for land uses, which increase the buffering capacity without changing the site quality. The minimum threshold eliminates variation. The various indicators deal with different components of the ecosystem, but they do not describe the ecosystem independently, so double counting is possible with the system, as is indicated by the PCA results. The PCA identifies which indicators are not necessary to assess and compare land use impact of subtropical agroforestry and mediterranean plantations with equal accuracy, but generalisation about indicator correlations are not possible. Assessing other land uses might lead to different indicator correlations. Conclusions of the method for criteria defined in section *2.2.* are presented in Table 3.

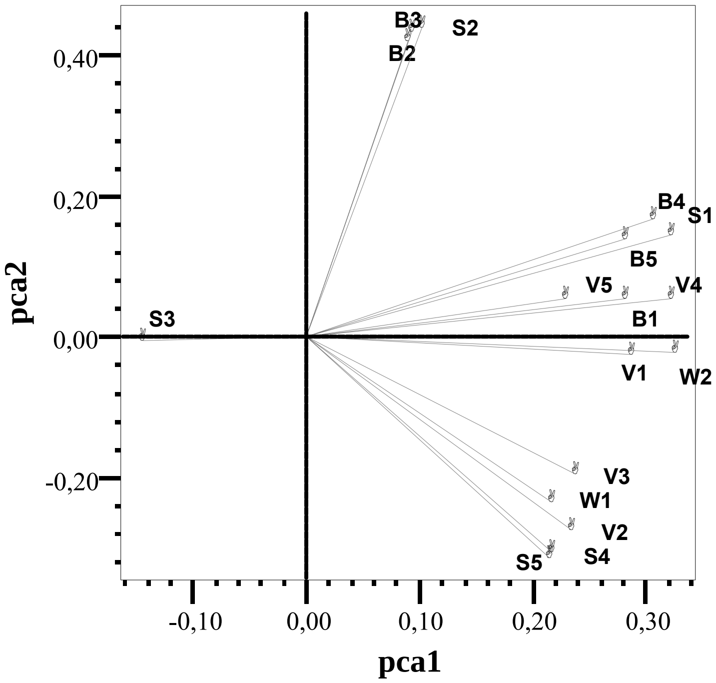


Figure 2. PCA of the MED pine plantation and SUBTROP agroforestry plantation scenario. Data labels are indicator codes (see Table 1)

The method does not aggregate theme scores into an overall score. This is interesting when LCA is used for decision-making. Production cycle 1 scores better under soil and water themes, while production scenario 2 scores better on vegetation and biodiversity for the land use category. To decide which is better, regional interests can be included. For example in South Africa where water is an important issue, production cycle 1 is preferred, while in Ireland where water availability is currently of little concern, production cycle 2 might be preferred because of biodiversity reasons.

Table 3. Results of the method for different criteria.

Legend: + = good, +/- = moderate, - = poor

criteria							
universal applicable	workable reference system definition	quantitative	use of functional unit	dimension	linearity	sensitivity	indicator efficiency
+	+	+	+	+	+	+/-	+/-

5. Recommendations

- Further testing of the method is needed. Special attention should be given to the indicator choices. Until now, many indicators seem to correlate, suggesting indicators could be eliminating to avoid double counting. Distinguishing between related land uses one might require all indicators.
- Indicators are designed as ratios between the actual -and reference states. All indicators are dimensionless, except for S2 and S3, which should be addressed.
- It is possible to add new indicators. A feasible indicator is based on red list species:

$$\left(1 - \frac{number\ of\ red\ list\ species_{act}}{number\ of\ red\ list\ species_{ref}}\right) * 100$$

This indicator represents nature value and, in exergy terms, it is important to conserve biodiversity as genetic information is a way to store exergy.

- Indicator minimum threshold needs revision. All indicators have a maximum of 100, but indicators do not have an intrinsic minimum. The –25 threshold was chosen to limit positive land use quality changes of man made land uses, but clearly affects sensitivity. Lowering the minimum threshold to –100 might solve sensitivity problems.
- The potential natural vegetation seems to be an excellent reference system, but problems can rise where no natural ecosystem is left for measurements. Literature data can be used but may affect overall reliability.
- Reference vegetation has the greatest possible exergy content for the site so impact should be 0, which is the case for all indicators except for soil erosion (S3) and surface runoff (W2). S2 and W2 should be adapted, so that reference vegetations score 0 on all themes.
- A general methodology to deal with uncertainty should be developed. The method as presented allows for comparison of many land use impacts, but reliability of assessment should be compared as well.
- Within themes the calculation (Eq. 1-4) of the impact scores gives equal weight to each indicator (e.g. in the soil theme soil erosion is as important as base saturation). This subjective weighting system needs reconsideration.
- Different themes cannot be compared within the same land use. All land uses have low soil and water scores, suggesting that soil and water indicators are less sensitive. Weighing factors can be attributed to indicators in order to make themes comparable, or themes themselves can be weighted. An important point is that different weighing systems should be used for different ecosystems but that objectivity and comparability of different land use assessments would be affected.

6. Acknowledgements

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7. References

Baitz, M., Kreissig, J. and Wolf, M., 2000. Methode zur Integration der Naturraum-Inanspruchnahme in Ökobilanzen. *Forstw Cbl* **119**: 128-149.

Bendoricchio, G. and Jørgensen, S.E., 1997. Exergy as a goal function of ecosystem dynamic. *Ecological modelling* 102: 5-15.

Bradley, M., Krokowski, K., Bresky, J., Pettersson, B., Woodtli, H., Nehm, F. and Daniel, P., 1998. Proposal for a new forestry assessment method in LCA. LCA graphic paper and print products (part 1). Axel Springer Verlag, Stora, Canfor study. 40 pp.

Content, T., 2003. Land use impact of a subtropical forestry plantation in Westfalia, South Africa: evaluation of a newly developed LCA method for impact category land use and comparison with land uses in different climatic regions. Diplom work, Laboratory for Forest, Nature and Landscape Research, KULeuven. 65pp.

Fath, B.D., Patten, B.C. and Choi, J.S., 2001. Complementary of ecological goal functions. *J. theor. Biol.* **208**: 493-506.

Giegrich, J. and Sturm, K., 1998. Operationalisierung der Wirkungskategorie Naturraumbeanspruchung. IFEU, Heidelberg. 19pp.

Giegrich, J., Schweinle, J., and Baitz, M., 1999. Verifizierung verschiedener Methoden zur Wirkungsabschätzung des Wirkkriteriums Naturrauminanspruchnahme, Landnutzung, Landverbrauch. Deutsche Gesellschaft fuer Holzforschung, Muenchen. 94 pp.

International Organisation for Standardisation, 1998. ISO/DIS 14042: Environmental management – Life cycle assessment – Life cycle impact assessment. Geneva.

Köllner, T., 2000. Species-pool effect potentials (SPEP) as a yardstick to evaluate land-use impacts on biodiversity. *Journal of cleaner production* **8**: 293-311.

Leplae, H., 2000. Comparison of different LCA methods for the impact category land use in a tropical moist evergreen forest (Ebom, Cameroon). Diplom work, Laboratory for Forest, Nature and Landscape Research, KULeuven. 89 pp.

Lindeijer, E.W., Van Kampen, M., Fraanje, P.J., Van Dobben, H.F., Nabuurs, G.J., Schouwenberg, E., Prins, A.H., Dankers, N. and Leopold, M.F., 1998. Biodiversity and life support indicators for land use impacts in LCA. Rijkswaterstaat, Delft. report W-DWW-98-059.

Lindeijer, E., Muller-Wenk, R. and Steen, B. (eds.), Baitz, M., Broers, J., Cederberg, C., Finnvelden, G., ten Houten, M., Köllner, T., Mattsson, B., May, J., Milà e Cannals, L., Renner, I. and Wiedema, B. (contributors), 2002. Impact assessment of resources and land use. Chapter in Udo de Haes, H.A. et al., 2002.

Luvall, J.C., Kay, J.J., Fraser, R.F., 2001. Thermal remote sensing and the thermodynamics of ecosystem development. In: Ulgiati, S., Brown, M.T., Giampietro, M., Herendeen, R.A., Mayumi, K. (Eds.), 2001. Advances in energy studies: exploring supplies, constraints, and strategies. SGE dictoriale, Padova. 147-158 pp.

Madueño Murillo, M., 2003. Land use impact assessment of Eucalyptus globulus in Spain. Diplom work, Laboratory of Forest, Nature and Landscape Research, KULeuven. 61 pp.

Muys, B. , Wagendorp, T. and Coppin, P., 2001. Ecosystem exergy as an indicator of land use impact in LCA. In: Ulgiati et al. Advances in energy studies: exploring supplies, constraints and strategies. Portovenere, Italy. pp 275-284.

Muys, B., Deckmyn, G., Moons, E., García Quijano, J., Proost, S. and Ceulemans, R., 2003. An integrated decision support tool for the prediction and evaluation of efficiency, environmental impact and total social cost of forestry projects in the framework of the Kyoto Protocol. International conference: Decision support for multiple purpose forestry, April 23-25, Vienna, Austria. 17 pp.

Oliver, M.A., 1997. Soil and human health: a review. *European Journal of Soil Science* 48: 573-592.

Peters, J., 2002. Comparative study of different LCA methods for the impact category land use in a Radiata pine plantation, Jonkershoek, South Africa. Diplom work Laboratory for Forest, Nature and Landscape Research, KULeuven. 106 pp.

Schneider, E.D. and Kay, J.J., 1994. Life as a manifestation of the Second law of Thermodynamics. *Mathematical and Computer Modelling* 19: 25-48.

Schweinle, J., 1999. Methode zur Integration des Aspektes der Flächennutzung in die Ökobilanzierung. In: Giegrich, J., Schweinle, J. and Baitz, M.. Verifizierung verschiedener Methoden zur Wirkungsabschätzung des Wirkkriteriums Naturrauminanspruchnahme, Landnutzung, Landverbrauch. Deutsche Gesellschaft fuer Holzforschung, Muenchen. 94 pp.

Schweinle, J., Doka, G., Hillier, W., Kaila, S., Köllner, T., Kreißig, J., Muys, B., Garcia Quijano, J., Salpakivi-Salomaa, P., Swan, G., and Wessman, H., 2002. The Assessment of Environmental Impacts caused by Land Use in the Life Cycle Assessment of Forestry and Forest Products - Guidelines, Hints and Recommendations. Final Report of Working Group 2 "Land use" of COST Action E9. Mitteilungen der Bundesforschungsanstalt für Forst-und Holzwirtschaft, Hamburg. 96 pp.

Scott, D.S., 2003. Exergy. *International journal of hydrogen energy* **28**: 369-375.

Sturm, K. and Westphal, C., 1996. Entwicklung einer Methode zur Bestimmung der Flächenqualitäten für Waldöksysteme für die Zwecke der Produkt-Ökobilanz. Büro für angewandte Waldökologie, Duvensee. 40 pp.

Udo de Haes, H.A., Jolliet, O., Finnvelden, G., Hauschild, M., Krewitt, W. and Müller-Wenk, R., 1999. Best available practice regarding impact categories and category indicators in life cycle impact assessment. *Int J LCA* 4(2): 66-74.

Soybean Production in Argentina: New Technologies, agricultural environment and socioeconomic impacts. The implications for the future

Walter A. Pengue
GEPAMA – www.gepama.com.ar, Grupo de Ecología del Paisaje y Medio Ambiente (Landscape Ecology and Environment Group), Universidad de Buenos Aires – Argentina, wapengue@gepama.com.ar

Introduction

Soybean has transformed itself into the most important crop of Argentina. In the same way, the country is at the global top ranking in relation with the adoption of the transgenic soybean technology. This year, planted surface with this crop rise to around 11,000,000 ha with a production of 34,000,000 metric tons (95% is transgenic).

For farmers, Round-up Ready (RR) soybean came to solve one of the main problems for the farm management: weed control, obtaining a cost reduction in the herbicide price, less fossil energy consumption and simple application that made the technical package offered irresistable. For the private pesticides and seed production sector, it opened the unique possibility to concentrate and rearrange the business of production and commercialization of insecticides and weed killers to the new biotechnological alternative.

The main area where soybean were produced was *The Pampas*, one of the most productive places in the world. But currently, due to the need for larger scale production, farmers are going out, increasing the pressure on more environmental sensitive areas. This situation is reproduced in Bolivia, Brazil and Paraguay.

The Pampas prairie is a vast, flat region of Argentina that comprises more that 50 million hectares of arable lands for crop and cattle production. Agriculture in the pampas has a short history (a little more than 100 years), and shared several common features with the agricultural history of the North American Great Plains. Both ecoregions were mostly native rangelands until the end of the 19th century and the beginning of the 20th, and both of them were later introduced into crop (cereal crops and oil seeds) and cattle production on dryland conditions.

The Pampas prairie is not homogeneous in soils (Morello, J and Matteucci, S, 1997). Using soils and rainfall patterns, the Pampas can be divided (Viglizzo, 2002) into five homogeneous areas: 1) Rolling Pampas, 2), Central Pampas (which could be subdivided in Suhumid on the East and Semiarid on the West, 3) Southern Pampas, 4), Flooding Pampas and 5) Mesopotamian Pampas.

In Argentina, specially in the Pampas, and now in areas out of it, from the north to the west, soybean production has, during last five years, displaced 4,600,000 ha dedicated earlier to other production, like dairy, fruticulture, horticulture, cattle or other agricultural sectors. More than fifty percent of the whole agrifood sector in Argentina (73,000,000 metric tons) will come this year from soybean sector. This unusual situation may endanger the stability of the Argentina economy or at least several sectors of it, as well as the food sovereignty of the country itself.

The increase of the soybean sector, which responds specifically to a global demand – Argentina consumes a very little of its own production – has produced important impacts on the environment, the economy and the society.

New technologies imported, success in economic terms of the No Tillage model, Transgenic Soybean and an explosive consumption of very specific pesticides have produced a particular combination that doubled the argentine production during the last decade: the "*Input Decade*".

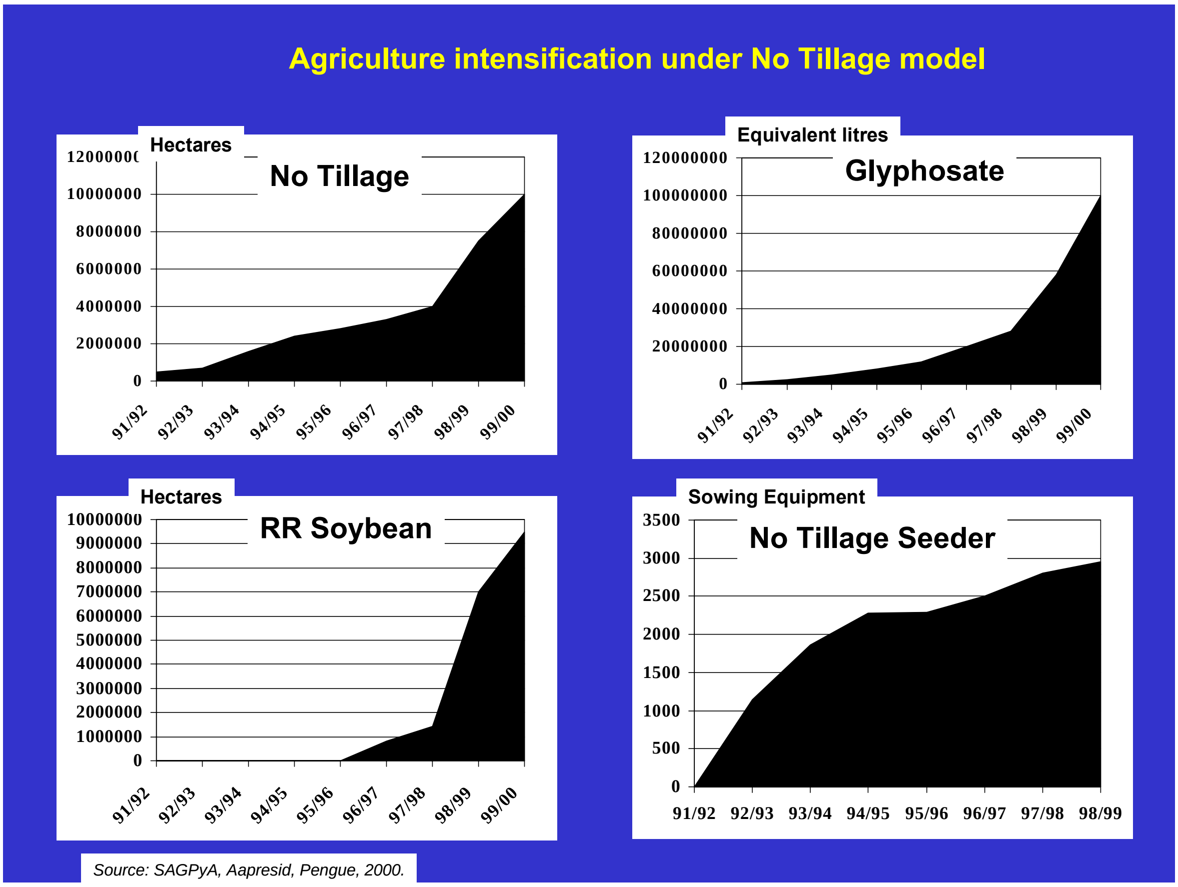
Landscape transformation in the rural sector is evident, the homogenization of the rural landscape and the transformation of virgin areas, could produce consequences that we are now evaluating, because this type of process has not a prior history in Argentina, neither in South American or global agriculture in general, and its impacts must be evaluated thoroughly.

Soybean Production in Argentina
Since 1997, the private companies, in many cases with the support and expectations of the government sector, has established that the transgenic package (Round- up Ready soybean + glyphosate) could offer to the country a real competitive advantage.

These advantages added to the already existing comparative advantages of the country could make Argentina one of the most efficient countries for producing and trading agricultural commodities.

In this way, since the 1996/1997 season, there has been a strong campaign for the commercialization of RRsoybean, that grew from 20% to 95% of the surface planted with GMO-soya in 2002/2003. In seven years, there was rapid adoptions of the new technology by growers, so that in the current season the whole production of argentine soybean is transgenic. Argentina did not generate the new technology, which has been imported by an international company's branch from USA.

Few years ago, traditional cultivation of grains alternated with fallow seasons for cattle pasture. This rotation system allowed maintaining the agronomic and environment system in the long-term. But, in the 1980s, world market prices for grains and oilseeds increased, while at the same time productivity of raising cattle declined. Agriculture became more lucrative, since the production of soybean in rotation with wheat or sunflower allows for three harvests in two year. Furthermore, the opening of the economy to the global market, the end of hyperinflation due to the fixation of the argentine peso against the US dollar, and abolition of export levies on agricultural products, triggered an investment in new technologies. This new framework favored the import of machinery and agricultural inputs as pesticides, fertilizers and royalties on seeds at low prices and their use in oilseed production under No Tillage system for export markets (Picture Nº 1).



Picture Nº 1. Agriculture Intensification under the No Tillage Model in Argentina. (RR soybean means GMO-soybean resistant to Glyphosate).

The intensification of the production system was followed by a decline in soil fertility and increase of soil erosion (Prego, E, 1996) Consequently, fertilizer consumption stepped up from 0.3 million tons in 1990 to 2.5 million tons in 1999. Another step was the continuos increase of No Tillage system that is directly associated with the high consumption of herbicides – such as glyphosate – reinforced with the release of transgenic soybean that are tolerant to this herbicide (package glyphosate + Roundup Ready soybean).

The main factors that produce the rapid adoption of transgenic soybean are:

a) Lower herbicide prices. In Argentina, from a price of $ 28/litre goes down now to $ 3/l, much less expensive that in the USA. Four companies (Monsanto, Atanor, Nidera and Dow) dominate more than 80% of glyphosate market in Argentina, mainly imported from USA, EU and China.

b) Fewer expenses on labor, fuel and machinery. No Tillage and more effective herbicide application allow for crop cultivation with less labor and fewer machinery cycles.
c) Complete knowledge of the technological package associated to No Tillage + Soybean.
d) Seed prices and self-reproduction. In Argentina, farmers don't pay technological fee for seeds and they reproduce the new seeds in theirs fields. This year the "*white bag*" (seed with no certificate and fiscalization) is around 300.000 ton.

Risks and profits under conditions of technological changing

Biotechnology is emerging at a period of worsening inequalities between the developing countries and the industrialized world. The income gap between the fifth of the world's people living in the richest countries and the fifth in the poorest was 74 to 1 in 1997, up from 60 to 1 in 1990 and 30 to 1 in 1960 (UNDP, 1999). And of the US$ 460 billion spent in R&D worldwide, only one tenth was spent in developing world where 80% of the world's population resides (UNESCO, 1999). These figures imply that much of developing countries are unlikely to benefit from biotechnology.

Although many of the developing countries are interested in the role of biotechnology for improving nutrition and reducing hunger, the majority of current agricultural biotechnology efforts are driven by the markets in the developed world: thus much of the research focus is on crops that are staple varieties for animal foods, attributes that minimize labor and comfort for farmers (unique herbicides, insecticides) or improve the quality of foods. Many of the crop varieties, traits and environmental or health conditions that could be important for large parts of the developing world are still largely ignored.

The intensification of agriculture implies for South American countries like Argentina and Brazil (the two main crop growers) two important transformations of land use:

An intensive production, under high input technology on common agricultural lands in the whole Pampas.

An extensive production, on new lands, gaining and advancing on marginal areas (agricultural border) with new varieties of soybean (transgenic and not transgenic), bred specifically to be adapted to these new virginal lands (Photo Nº 1).

Photo Nº 1. Clearfield of lands in the Yungas forest prepared for soybean production (Salta, Argentina, 2002).

Three decades before, soybean was a botanical curiosity. Nowadays, it is the engine of MERCOSUR. It is the third exportation good (after coffee and sugar) and the first of Argentina. But, both countries have followed different goals and different views of markets. While Argentine followed the United States and continue with the intensification of OGM production, the production and release of engineered crops is under discussion in Brazil, where the government only allowed planted transgenic soybean this season, with an open end for the next future (Table Nº 1).

Table Nº 1. Growing of surface implanted with soybean in Argentina and Brazil.

Country decade	1970	1980	1990	2000
Argentina Hectares	50,000	2,000,000	5,000,000	8,000,000
Brazil Hectares	1,000,000	6,000,000	12,000,000	13,000,000

Source: Pengue, Walter. Seminar Sustainable Agriculture in the Third World, pp.71-87 (2002), Brussels.

The current situation seems to be a bifurcation of the world market. By one side, those countries that accept engineered crops and those that do not accept engineered crops or insist that those crops and foods have to be labeled.

Current situation and trends for agriculture production in Argentina.
Argentina can be considered as a "*natural*" country, free till the first years of this decade of high inputs of chemicals as fertilizers, insecticides or herbicides for its crops (Table Nº 2). This is a "*market value*". But, in hands of globalization and facing an important soybean demand, the country is changing its system of production, intensifying agriculture, with high consumption of imported chemicals, new varieties of crops and a class of agricultural biotechnology that implies more consumption of herbicides, with active principles [ingredients?] imported too.

Table Nº 2. Some agriculture indicators in selection agriculture economies.

	Argentina	USA	France
Insecticides (gr./ha)	250	1000	3000
Fertilizer (kg. /ha)	25	100	300
Herbicides (gr./ha)	250	900	2000
Changes in farm area (%)	18	5	- 2.5
Native mammals under danger of extinction (%)	10	11	50
Native birds under danger of extinction (%)	2	8	40
Native reptiles under danger of extinction (%)	0	6	38

Sources: INTA (1995), INRA(1995), USDA (1996), Pengue (1996)

Historically, Argentina has been characterized for its natural conditions, that even following the intensification of the "*green revolution*", the country did not consume much chemicals. Only the erosion and nutrient exportation have been important as a consequence of wrong management and the incorporation of the package for soybean, without the right evaluation of the environmental context. But nowadays, adding to the problems with the soil resource, the entire ecosystem will be involved. The "*new biorevolution*", in the way that is being promoted in Argentina, will allow increasing the agricultural cycles, diminishing the length of fallow fields and restoration, increasing the impacts and pressure over natural resources, the social system and the economy.

Agriculture intensification has produced environmental, economical and social consequences that have not been evaluated conspicuity in the country. Probably, the new *biorevolution* could exacerbate the weak conditions of the system: Intensification of agriculture, globalization, large farm concentration, low levels of credit for small farmers, dependence of imported supplies, dependence of technology, apropiation of large farms by outside owners, concentration of seeds and chemicals on a very few agricultural firms.

This simplification of agriculture will produce effects that will affect the commercial position of Argentina in the meantime: degradation of soils and biodiversity, rural migration, concentration in large farms only producing high yielding crops in place of more natural foods.

There are social and economic consequences related with the important changes and transformation of national economy. Since 1991, starting the period of dollar convertibility and opening of the argentine market, changes in the mode of production have led to a number of social transformations for the agricultural sector:

a) Dependence on imports. Grains and soybean have become the main goods for foreign markets, boosting the dependence on import of the inputs. Local production of pesticides rose 16.6%, while 43.6% are imported and the other 39.8% are produced in Argentina with imported drugs. Glyphosate consumed in 2003 is around 160,000,000 lt.
b) Concentration of holdings. New technological package offered in a context of profit margins falling down by half between 1992 and 1999, makes it very difficult to survive for many farmers indebted with bank loans of high interest rates to pay back for these investments in machinery, chemical inputs and seeds. This situation favors the concentration of holdings and many farmers (especially small and medium size growers which were the train of the argentine economy) disappeared. Between 1992 and 1999, the number of farms in Las Pampas declined from 170,000 to 116,000, while the average size of a producer's farm increased from 243 to 357 hectares. In 2003: 532 hectares.
c) Dumping prices. Argentina as many developing countries subsidize neither its farmers nor the goods they produce, but are being affected by those governments that subsidize the production of commodities in developed countries. In this way, these activities promote an intensification of agriculture production in developing countries, overexploitation of resources and subutilization of goods (that excluded the valuation of externalities).
d) Exclusion of small farmers, who cannot get financial support, for the acquisition of the technological package.
e) Adverse consequences for organic farming by contamination or gene flow.

It is a real consideration that short-term economic and social objectives that ignore mid and long-term environmental effects put the future sustainability of the society at risk. However, although indicators to measure social or economic changes are abundant, indicators for assessing environmental changes are scarce. The generation and development of proper indicators for an agro-environmental information system are essential to get a permanent quality assessment of rural environments.

About the current situation and the exploitation of the environment under a typical situation of pressure and technological change we can question if our Pampas are sustainable at this time? Where were we and where are we in environmental terms?, What are the tendencies?, Which are the most worrying ones?, and which are the most appropriate indicators for an encompassing evaluation of the environment of the Pampas today. Another question is how these indicators are related to the social and economical ones mentioned before.
The first results are available from studies that evaluated twelve indicators: Land use, consumption of fossil energy, fossil energy use efficiency, nitrogen and phosphorus balances, ni-

trogen and phosphorus risk, pesticide contamination risk, relative levels of habitat intervention, changes in Carbon stock and greenhouse gases balance.

Land use is the most important factor that drives the environmental behavior of the region (Table Nº 3). All indicators, from fossil energy consumption to contamination risk, from erosion risk to greenhouse gases emission, are particularly sensitive to land use. Technology is the next factor (Viglizzo, op. cit).

Table Nº 3. Changes in the area allocated to predominant annual crops (soybean, maize, wheat and sunflower) in the Argentina Pampas during the period 1960-2000.

Percentage of the Total Area

Pampas Area / Year	1960	1988	1996
Regional Average	**23.70**	**30.30**	**40.00**
Rolling Pampas	**28.90**	**47.60**	**63.40**
Central Subhumid	31.30	38.30	53.60
Central Semiarid	21.70	38.40	39.10
Southern Pampas	23.40	32.40	36.80
Flooding Pampas	12.20	8.20	13.20
Mesopotamian Pampas	10.40	7.60	10.40

Source: National Program of Agro-Environmental Management, Argentina, 2002.

Trends in fossil energy consumption in The Pampas indicate that intensification is increasing at high rates. Land productivity and fossil energy consumption have almost doubled in less than ten years. This shows a direction of Argentinean agriculture towards a more intensive model, departing from the traditional semi-intensive one. Rolling Pampas has an energy budget that highly exceeds the whole Pampas average (Table Nº 4).

Table Nº 4. Energy productivity and fossil energy consumption in the Argentine Pampas during the period 1960-2000. Comparison of trends among ecologically homogeneous areas.

	Energy Productivity (Gj/ha/year)			**Fossil energy consumption (Gj/ha/year)**		
	1960	1988	1996	*1960*	*1988*	*1996*
Regional Area	6.40	13.45	22.16	*1.30*	*1.68*	*3.31*
Rolling Pampas	**9.03**	**24.11**	**31.92**	***1.27***	***1.95***	***3.79***
Central Subhumid	6.39	14.40	25.59	*1.88*	*2.00*	*3.81*
Central Semiarid	2.75	4.33	8.43	*1.19*	*1.88*	*2.68*
Southern	5.44	11.23	19.48	*1.15*	*1.78*	*3.12*
Flooding Pampas	3.48	4.19	10.91	*0.56*	*0.50*	*1.43*
Mesopotamian	3.21	3.42	13.92	*0.56*	*0.51*	*1.86*

Source: National Program of Agro-Environmental Management, Argentina, 2002.

In a general context, the Pampas has not been fertilized till the beginning of the nineties. The nutrient budget of the Pampas had some stabilization before this time, by the rotation of crops and cattle, the most common production system in the area. But it was in the nineties when the land use transformations and an increase in fertilizer use drove the *Argentinean Pampas* into more intensive models that are typical of the northern hemisphere.

Soybean has had and will have an emblematic role in relation with nutrient balance, loss of quality and richness of ours soils.

Each year the country exports with its grains a considerable amount of nutrients – especially nitrogen, phosphorus and potassium – that in the process of intensification, are not replenished. Argentina exports yearly around 3,500,000 metric tons of nutrients – with no recognition in the market prices, increasing the "ecological debt" (Martinez Alier and Oliveras, 2003). Soybean, the engine of this transformation, represents around fifty percent of this average. If we compensate the natural depletion with mineral fertilizers, Argentina will need around 3,326,786 metric tons of nitrogen and phosphorous fertilizers and an amount of 900,000,000 American dollars to buy it in the market (Table Nº 5) (Pengue, W, 2003).

Table Nº 5. Stimation of nutrients (N, P) exportation and the cost for soybean harvest 2002/2003 (34,000,000 metric tons).

	Nitrogen	Phosphorous	Total
Nutrient Extraction in metric tons	1,020,000	227,800	1,247,800
Equivalent in Mineral Fertilizers in metric tons	2,217,400	1,109,386	3,326,786
Cost Stimation reposition (US$)	576,524,000	332,816,000	909,340,000

Source: Pengue, W. La economía y los subsidios ambientales: Una Deuda Ecológica en la Pampa Argentina. Fronteras Nº 2: 7-8. Año 2. Number 2. GEPAMA.FADU.UBA. Buenos Aires. 2003.

Stimation for next season (2003/2004) considered that around 30% of the whole soybean area (4,500,000 hectares) will be fertilized with mineral fertilizers. In 2002/2003 surface implanted with soybean rose to 12,900,000 ha and next season the estimation is around 13,600.000 hectares. The scenery shows a trend in important depletion of nutrients in ours soils that will be consumed completely in 50 years (Ventimiglia, 2003).

Under No Tillage system, indicators show that soil erosion risk tend to decline. Although the area cultivated with annual crops expanded, minimum and No Tillage practices compensated the more intensive use of land.

The use of land and the expansion of the agricultural border is the most relevant impact factor after losing biodiversity. Aggressive agronomic practices, global demand for soybean and attractive prices without institutional national regulations and economic instruments, reinforce the negative impact of intensive land use on habitats and biodiversity. Argentina is one of the countries in South America that posses less territory as protected area (4.8% of 2,777,815 km2; others countries: Brazil 16.8%, Bolivia 22.4%, Peru 9.9%) (Burkart, 1999)

Final Comments

The intensification of Argentine agriculture, represented by the use of No-tillage practice and glyphosate, has allowed for the homogenization of production based on transgenic soybean as the dominant crop.

The present export-oriented commodity production system is most likely to drive more smaller farmers out of business. For them, a diversification beyond global commodity markets, be they non-transgenic for export or other crops for internal purposes, might render an alternative development trajectory. However, this would require a drastic turn in Argentine's agricultural policy, namely to play a more active role and to subsidize small-scale farmers.

The overwhelming domination of transgenic soybean makes farmers especially vulnerable to changes in the global commodity markets due to the preoccupation regarding the safety of GMOs.

About environment, first indicators show interesting impacts, some of them directly related to soybean production and intensification practices. The transition towards a more intensive model of agricultural production, both in terms of land use and technology application, has characterized the nineties. Many farming systems in Argentina resemble some intensive models that are very common in the United States or Europe.

Nutrients depletion is a new complex discussion that must be solved with holistic agroproductive policies, not with current decisions as increasing of application mineral fertilizers.

Deforestation and expansion of agricultural borders must be analyzed fast and need policy decisions to avoid an important loss of biodiversity and habitat.

Indicators showing a negative trend are important keys to identify critical problems that will require a more specific attention of researchers and government. An increase in contamination risks, fertilizer supplies, lost of biodiversity, and monoproduction must to be discussed to assure the future sustainability of *the Pampas*.

References

Brodnig, G. Biotechnology in international trade. Weatherhead Center for International Affairs. Harvard University, Set. 1999.

Burkart, R. Conservación de la biodiversidad en bosques naturales productivos del subtrópico argentino in Matteucci, S and Solbrig, O. Comp. Biodiversidad y Uso de la tierra. Eudeba. Unesco. Buenos Aires, 1999.

CID – Center for International Development. Agricultural Research in Africa: Technological opportunities and institutional challenges: Report of a Seminar. Center for International Development. Harvard University. 1999.

James, C. The global review of commercialized transgenic crops. ISAAA, Brief paper – 2000, Ithaca, N Y, USA. 2000.

Lehman, V and Pengue, W. Herbicide tolerant soybean: Just another step in a technology treadmill? Biotechnology and Development Monitor Nº 43, September 2000.

Martinez Alier, J and Oliveras, A. Deuda Ecológica y Deuda Externa. Quién debe a quién?. Icaria. Barcelona, 2003.

Morello, J et al. Argentina, granero del mundo ¿Hasta cuando?. Orientación gráfica Editora. Bs. As. 1997.

Morello, J y Matteucci, S. Estado actual del subsistema ecológico del nucleo maicero de la Pampa Húmeda. Capitulo Nº 4. En Argentina, Granero del Mundo, ¿Hasta cuándo?. Orientación Gráfica Editora. Buenos Aires. 1997.

Pengue, W. The agriculture's sustainability in Argentina. The fourth biennial meeting of ISEE, Boston University. 1996.

Pengue, W. Sojas Transgénicas: Tecnología y Mercados. Realidad Económica. IADE. Buenos Aires, Mayo, 1999.

Pengue, W. Cultivos Transgénicos ¿Hacia dónde vamos? Lugar Editorial. UNESCO, 208 páginas. Buenos Aires. 2000.

Pengue, W. Commoditación o diversificación de la producción agricola argentina. Transgénicos. Jornada de Biotecnología en el agro. Universidad Nacional de La Plata. 2000.

Pengue, W. Impactos tecnológicos y ambientales de la liberación de OGMs. Conferencia Internacional sobre Comercio, Ambiente y Desarrollo Sustentable. Programa de las Naciones Unidas para el Medio Ambiente. PNUMA/UNEP Mexico, Febrero, 2001.

Prego, J et. al. El deterioro del ambiente en la Argentina (suelo-agua-vegetación-fauna). 3º Edición. Fundación para la Educación, la ciencia y la cultura. Buenos Aires, 1997.

Sachs, J. The Economist. 1999.

UNDP. United Nations Development Programme. Human development reports 1999: Globalization with a human face. UNDP, New York. 1999.

UNESCO – United Nations Scientific, Educational, Scientific and Cultural Organization. World Science Report. Paris. 1998.

Ventimiglia, L. El suelo, una caja de ahorros que puede quedar sin fondos. La Nación. Suplemento Campo. Pag. 7. Buenos Aires, October 18th. 2003

Viglizzo, E. F et al. La sustentabilidad ambiental del agro pampeano. Ediciones INTA. Buenos Aires, 2002.

How may Quality Assurance Systems in food chains include environmental aspects based on Life Cycle Methodology?

Niels Halberg

Danish Institute of Agricultural Sciences, Department of Agroecology, Research Centre Foulum, P.O. Box 50, DK-8830 Tjele. Tel: +45 8999 1206 - Fax: +45 8999 1200. E-mail: Niels.Halberg@agrsci.dk

Abstract:

The number of Quality Assurance Systems (QAS) for food products is increasing and the so is the topics they cover, from traditional intrinsic product characteristics such as percent meat in slaughtered pigs and protein content in milk to food safety issues such as zoonoses and pesticide residues and in some cases aspects of animal welfare. This development is linked to demands for risk controlling systems such as HACCP and traceability systems that would allow food safety problems to be traced to a small number of producers or farms. The large retail companies (supermarkets) are an important driving force for this development because of their efforts to build consumer trust in food products and loyalty to the companies own brands. Environmental characteristics of food products and information on their production methods are becoming part of some QAS but not mostly in the form of qualitative information e.g. certification that the farmers have used Good Agricultural Practice (GAP). The paper gives examples of this and then discuss this development in relation to LCA based environmental appraisal of food products. The development of quantitative (tools for) environmental appraisal of agriculture and food production is becoming more productoriented improving the possibilities of assessing the regional and global impacts of food production chains and consumption. But these systems building on LCA does not so far seem to be linked with the development of QAS for food. The paper finally discuss the possibilities for linking the food safety related traceability systems and gives an example of on-going work to establish LCA based QAS in a meat processing system.

Introduction

In the wake of the great European food scares of the 1990'ties a number of Quality Assurance Systemes (QAS) have been introduced or improved by food processing and retail companies in order to increase and regain consumers' confidence in food products (Schiefer, 2004) and to secure against liability from unforeseen food hazards. The more elaborate QAS build on a number of safety and control measures at critical points in the production process to avoid contamination and spread of food hazards. To secure transparency there is often an intensive information flow and assurance of tracking and tracing of product components between different steps in the product chain. Some slaughterhouses, for example, keep track of the meat quality from each primary producer thus keeping records that allow to track batches of meat from supermaket back to a very small number of farms. This increased interest in quality control and traceability has only to a limited extent included externalities of the production such as the animal welfare and environmental impacts through the production chain even though these may be considered as equally important attributes of food products from a societal point of view.

Consumers appreciate a number of different quality aspects of food products among which are both intrinsic and extrinsic. Following Steenkamp (1990) the intrinsic characteristics include the organo-leptic or sensoric quality such as colour, taste, visible fat (called "quality cues" if they are observable before purchase and "experience attributes" if they may first be ascertained while consuming the product). Intrinsic characteristics that relate to food safety may often not be revealed immediately ("credence attributes") and it is off course no surprise that many consumers have become interested in information regarding the risk of zoonoses, bacteria or other food hazards when choosing food products. Extrinsic characteristics – besides price - relate to the conditions of livestock in the production chain (animal welfare), to the resource use and environmental impact from the production and to other aspects not observable from the product itself (e.g. GMO free, organic production, regional product)(Brom, 2000; Verbeke and Viane, 2000).

It has been proposed that the environmental information from e.g. Life Cycle Assessment (LCA) may be used to guide consumer choices (Nilsson et al., 2004) at least in public procurement (Anonymous, 2002). The EU Integrated Product Policy (IPP) considers LCA as one of the cornerstones (Anonymous, 2003) for improvement of the knowledge and transperancy concerning the environmental impacts related to production and consumption. However, it is not clear how significant environmental labelling is for consumers' preferences and other aspects of food quality are probably more important (Brom, 2000; Verbeke and Viane, 2000; Nilsson et al., 2004). It seems as if presently environmental aspects has less priority compared with food safety issues in the development of traceability and documentation in QAS (maybe because food contamination can have direct impact on specific consumers while most environmental issues have a less direct impact on particular consumers using a specific product). But in a larger context the environmental characteristics of food production chains are important because the food production and consumption is one of the larger contributors to a family's environmental impact (Anonymous, 1996; Wilting et al., 1999; Spangenberg & Lorek, 2002). Livestock products are particularly important for the emissions of nutrients and greenhouse gasses (see several papers in this volume) and for land use (Gerbens-Leenes et al., 2002) and projections of global food demands and production foresee a significant global rise in consumption of meat and milk (Delgado et al., 1999). Therefore, it seems relevant to seek ways to improve environmental appraisal of agriculture and food chains with the aim of reducing the environmental load per kg product produced and consumed.

There are examples of larger food companies - such as Arla, Unilever and Cerelia - performing LCA on specific projects (Larsson, 2004; McKeown, 2001; Rosing et al., 2004) either as part of their product development or in order to be prepared against criticism from environmentally conscious consumer groups. Also, a number of food processing plants like slaughterhouses and dairies have used energy accounting tools or other environmental management tools and some have become ISO 14001 certified such as the Danish slaughterhouse “Tican”. However, often the most significant environmental impacts from food production are happening in the primary production rather than the processing stages. Therefore, attempts to describe, appraise and document environmental characteristics of food products should include the whole production chain. A number of European tools for farm level environmental appraisal and reporting exist for voluntary use, some of which are linked to advisory tools for farm planning (Halberg et al., 2004). But these tools are seldom linked with the rest of the food chain and existing labels for environmentally friendly production are usually not based on quantitative information of potential environmental impact from the specific producers (Nilsson et al., 2004).

The aim of this paper is therefore

- to discuss trends in Quality Assurance Systems (QAS) used in European food chains and the different methods used for appraisal of environmental characteristics of food products
- to propose that environmental appraisal as part of QAS should be based on quantified information through the product chain and be linked to traceability principles.

Quality Assurance Systems in European food sector

A number of QAS for food products exist in Europe and other countries differing in both their organisational set-up and in the degree to which they build on international certification standards (Schiefer, 2004). Meat and animal products have been at the core of food scandals in the last decade and not surprisingly some of the strongest QAS are to be found within pork production, such as the Dutch IKB (Trijp et al., 1997), a relatively new German system called QS (Quality and Safety for Food Products, from Producer to Consumer" (Nienhoff, 2004; www.q-s.info) and the “Danish” brand (Anonymous, 2003). These management schemes aim at securing a high degree of traceability through a strong information flow where results from tests and measurements along the production chain are continuously fed backwards with the aim of improving performance (e.g. feeding back information to farmers on carcass quality or bacterial counts in milk). Most often this relates to analyses of ”classical” quality parameters such as fat content, freezing point and bacterial counts and for medicine residues and other more recent tests such as Aflatoxines. Thus, most often QAS builds on two key concepts: Traceability and HACCP. HACCP (Hazard Analysis and Critical Control Points (CFSAN, 2004; Danske Slagterier, 2003) is a systematic approach to monitoring quality and risks of contamination of food products during a production process and is standardised among others in Denmark and Holland and internationally in ISO 22000.

Moreover, increasingly the QAS seek to include whole product chains in order to improve cross-border trade and share information not only backwards/upstream but also downstream in the chain as documentation coming with the product. However, when it comes to information regarding the extrinsic characteristics such as animal welfare and environmental performance such information is presently not accumulated and exchanged through the value chain in a quantitative form. The QS system for example includes demand for compliance with other German regulation of manure use but no quantification of nutrient surplus or losses. Likewise, the French QAS "Agri Confiance" for certification of a variety of livestock products and processed crop products has only recently started to include environmental aspects in a separately managed ISO 14001 scheme called Agri Confidence® Quality-Environment (www.cooperation-agricole.asso.fr cit. Cederberg, 2004). The intention of labels such as IKB are to add value to a product (pork) in the eye of the consumer by guaranteeing that it is produced according to some criteria, which are believed to be important (Trijp et al., 1997). This added value should then give an advantage in the form of either higher prices or increased consumer loyalty to products labelled with the IKB brand.

In addition to the brands and QAS of the food processing industry large retail companies also begin to focus on traceability and documentation of the origin and quality of the products they sell. There seems to be a trend towards non-price competition (competing on the products' quality attributes) and building consumer loyalty towards the retail companies' own brands. An increasing proportion of the retail sale of food and household commodities happens under the retail companies' own brands (Arfini and Mancini, 2004). The British Tesco and Sainsbury for example both have their own brands that account for over 50% of their total sales, the French Carrefour and Intermarche sell 20 and 29% of their turnover in their own brands and this proportion is increasing also in the Nordic COOP chain. This has economical advantages for the retail business and gives them stronger control and flexibility (e.g. they may change their suppliers without the consumers noticing). To minimise the risk of loosing consumer confidence in these trademarks due to food scandals the retail business is now very active in quality assurance and therefore demand quality control measures and traceability backwards in the food chain.

Arfini and Manicini (2004) studied the British Retail Consortium (BRC) as an example of this involvement. The BRC is an association of major retail chains and distribution companies in the UK and has as its major function to translate consumer demands and interests into demands for the products' characteristics and performance through the supply chain. Thus, the BRC's so-called "Technical standard and protocol for companies supplying retailer branded food products" include demands that companies establish hygiene and safety control systems based on the HACCP method. Suppliers should also adopt a documented quality management system including requirements of minimum levels and recommendations on good practice, following a standard such as EN45011, which is basically in compliance with ISO 9001. This way (expected) consumer demands concerning food safety and product quality have been translated into contractual requirements that suppliers of food products (also foreign) have to comply with when dealing with the members of BRC, which is the majority of the large retail chains in the UK.

According to Krieger and Schiefer (2004) the primary agricultural production (e.g. the pig fattening facility) will become integrated in HACCP systems in the future and HACCP systems already exist for fruits and vegetables (Hernandez-Souchez et al., 2004). However, the authors do not find it likely that these quality assurance schemes will be used to claim higher prices for the certified products. Rather these concepts will be considered the standard or basic quality for a number of food items. Contrary to this Broom (2000) argues that food safety may be assured by labelling schemes if they are backed by government control systems. Other issues that are not necessarily relevant to all consumers in their role as consumers - such as animal welfare or environmental issues – may also be assured by labelling. The European Commissions study on "environmental product declaration schemes" (Anonymous, 2002) advise that information on the environmental performance of food products be considered in e.g. public procurement. Nilsson et al. (2004) find that existing labelling schemes often lack credibility in the sense that "they are perceived interesting and trusted by consumers" and should therefore be backed by a more factual appraisal of the actual production methods and their environmental impact. Whether the QAS will be used to give information to consumers or will remain primarily a process between agents in the food processing chain is not clear, but in both cases it seems relevant to discuss the potential role of LCA or other forms of environmental assessment to supplement such systems.

Good Agricultural Practice and environmental Quality management of food products
Some initiatives aim at securing minimum standards for the environmental performance of agricultural products, especially through the establishment of certification schemes[1]. A major objective among retail companies is to avoid pesticide scandals caused by either too high amounts of residues in the products or hazardous use by the farmers. This is primarily secured by either banning certain pesticides in specific products or demanding rules for pesticide use and storage, see below. Other aspects are included such as the prudent use of water and fertiliser but most often in a non-quantitative form, which seem less rigorous than for the above mentioned meat quality and food safety issues. Therefore, it is relevant to discuss the possibility of using quantified information based on actual use of inputs and/or estimated emissions in the environmental assessment of food products.

A number of food products are produced and sold under labels claiming some form of environmental consideration. One example is the bread wheat and rye sold under the *NATUR+* label

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[1] Certified organic farming is off course also a certification that certain practices have been followed and that no pesticides have been used but it does not quantify the environmental impact as such and will not be considered here.

(Cederberg, 2004; Swedish Seal, 2004) owned by the Swedish farmers and used for most bread and flour sold in Scandinavian supermarkets such as COOP in Denmark and Sweden. The *NATUR+* label guarantees that no chemical "plant growth regulators" have been used (following specific worries for food safety of this otherwise legal crop treatment) and the rules also ban pre-harvest Round-up use and use of sewage sludge. It may be discussed whether these rules in reality address a consumer concern for chemical residues in the bread rather than care for the environment, but recently also rules for fertiliser planning and minimum requirements for "green zones" on the farm have been included.

Under the British Farm Standard logo of a "Little Red Tractor" exists a number of guidelines for assured production with rules of how farmers should take environmental considerations in their planning and management. One example is the Assured Produce Scheme (APS), which promotes safe and environmentally responsible production of fruit, salads and vegetables through the use of integrated crop management (ICM). According to the home page (Anonymous, 1993) APS "is designed to maintain consumers' confidence in the safety and integrity of the produce they eat". Growers must follow the best production advice contained in the crop specific protocols that form the basis of the scheme. For example, the use of fertiliser should be based on crop norms and soil analyses and pesticides should only be used after observation of a critical level of a pest in the crops. APS thus follows a logic of Good Agricultural Practice (GAP) and is crop/field level based, not product oriented. APS is an independently assessed assurance scheme and farmers have to be certified and inspected to sell products labelled with the Little Red Tractor. Other QAS under the little red tractor cover e.g. pigs (Assured British Pigs) and chicken (Assured Chicken Production).

The APS is one of many examples of a labelling scheme certified under the umbrella EurepGAP, which is an initiative owned by a consortium of European retail companies (supermarket chains). The EurepGAP is based on the socalled *FoodPLUS / STATUTES which have the objectives to:"Encourage adoption of commercially viable Farm Assurance Schemes, which promotes the minimisation of agrochemical inputs, within Europe and world wide.*
Develop a Good Agricultural Practice (GAP) Framework for benchmarking existing Farm Assurance Schemes and Standards including traceability." (EurepGAP, 2003).

The Danish IP label (Integrated Production) for vegetables (outdoor as well as greenhouse crops) is owned by an independent group of horticulturalists organised under producer organisations (GAU/DEG/GASA) (Anonymous, 2004a). Danish IP is based on the idea of promoting the use of good crop rotations and other preventive measures to reduce the need for pesticides as much as possible. As an example the producers of IP tomatoes and cucumbers have to record and document that they purchase and use biological control of pests. It was originally the hope among the initiating producers that the IP label would qualify for a price premium but this has not been realized. However, the Danish IP label is credited for the relatively high proportion of Danish produced vegetables sold in supermarkets. The Danish IP is currently undergoing adjustments to comply with the EurepGAP standards. This implies some changes in the level of documentation and in specific rules for e.g. storage of pesticides but not in the actual environmental performance of the farms. This is because the EurepGAP standards do not include specific quantified limits for e.g. fertiliser use or environmental impact.

Environmental assessment as part of quality assurance schemes

Why should the food business use precise documentation of environmental impacts of food products? There are a least two reason why documentation of environmental characteristics should be included in QAS. The first reason is related to the interests of the brands and food companies and the second is related to the societal interests in environmental improvements in the food chain.

1. The advantages for the food companies of branding builds partly on the ideas that the perceived better quality associated with the brand and other brand associations increases the consumers experience and satisfaction with a product (van Trijp et al., 1997). As discussed in the introduction "quality" is more than the intrinsic characteristics of a product and includes external characteristics such as environment and animal welfare aspects of the production process. Therefore, it may be an advantage towards at least some parts of consumers to have documentation that environmental care is part of the brand policy. Or, at least the retail companies and the brand owners should try to avoid critical stories concerning the environmental impact from their suppliers, e.g. damaging losses of critical pesticides.
2. From a broader environmental perspective there could be an advantage of including environmental assessment in QAS because important environmental impacts from food consumption are regional and global such as nutrient losses and greenhouse gas emissions. The larger part of these environmental impacts happen in the primary production and there seems to be a potential for improvement as demonstrated by both LCA studies (de Boer, 2003; Haas et al., 2000; Halberg, 1999; Erzinger, 2004) and the tests of farm level green accounts (Halberg et al. in press). To record and report environmental information would facilitate better control, regulation and improvements based on incentives from e.g. the retail companies with their own brands. Following the ideas of the IPP information regarding the environmental impacts accumulated through the food processing chain would also facilitate better choices among retail companies and professionals in kitchens and restaurants.

As mentioned above, quality parameters like carcass quality is communicated up- and downstream in the food chain partly as quantitative information but the environmental information - if used at all - is based mostly on GAP, adherence to decision rules etc., not on the actual result, e.g. resources used per kg product or LCA type of information. It should however be possible to use quantified environmental assessments because this type of information is the baseline of many types of green accounts for farms in Europe (Halberg et al., 2004), but these are most often not integrated with the product chain QAS. Nilsson et al. (2004) analysed the credibility of 58 eco-labelling schemes and conclude that presently ”There is no labelling system that covers the entire food production chain which could install ecoefficiency in the production chain”. The authors call for an “alternative approach [that] could measure appointed quality aspects in indicators for the whole food product chain and report them to interested parties and consumers” (Nilsson et al., 2004).

Some of the objectives for including environmental characteristics in QAS may be fulfilled by GAP type rules to be followed by the farmers, e.g. certification that they only use legal pesticides and only after pest-infections above a certain threshold have been observed by the farmer in his fields (such as is the generic rules in EurepGAP standards). From the point of view of the retail sector this approach seems to limit the risk of food scandals caused by misuse or overuse of pesticides by the primary producers. They may claim that they have done their best to limit this and thus avoid liability in case of such a case becoming public.

However, this rule based method for environmental QA does not significantly document any improvements in environmental performance compared with standard practice, especially not in countries with a high standard for public regulation. Rules for GAP that simply demand the farmer to make a fertiliser plan or to use pesticides only after inspection in the crops do not distinguish between farmers who use only small amount of pesticides or fertilisers (e.g. because they have a better crop rotation) and those who in reality rely on standard dosages. Halberg et al. (2004) compared different European concepts for farm level environmental appraisal (Input output accounting, Green Accounts etc.) and found a similar distinction between management (rule) based indicators and quantitative indicators based on results of environmental performance on farms (e.g. nutrient surplus per ha or energy use per kg product). It was concluded that the results based indicators were more suitable to link with advisory tools for improving farm performance based on e.g. benchmarking. Benchmarking is here understood as “the process of improving performance by continuously identifying, understanding and adapting outstanding practices and processes found inside and outside the organisation” (Amer. Prod & Quality Center, 1999, cit. EEA, 2001). In other words, benchmarking is to compare one’s own results with other producers’ performance and thereby identifying ”best practices” among comparable producers. The process also involves the tasks of understanding these differences, thus learning from others and using this to set goals for one self and the engage in activities to improve one’s own practices. To perform benchmarking and facilitate improvements there is a need for quantitative assessments of environmental characteristics of food products based on the actual processes and resource use.

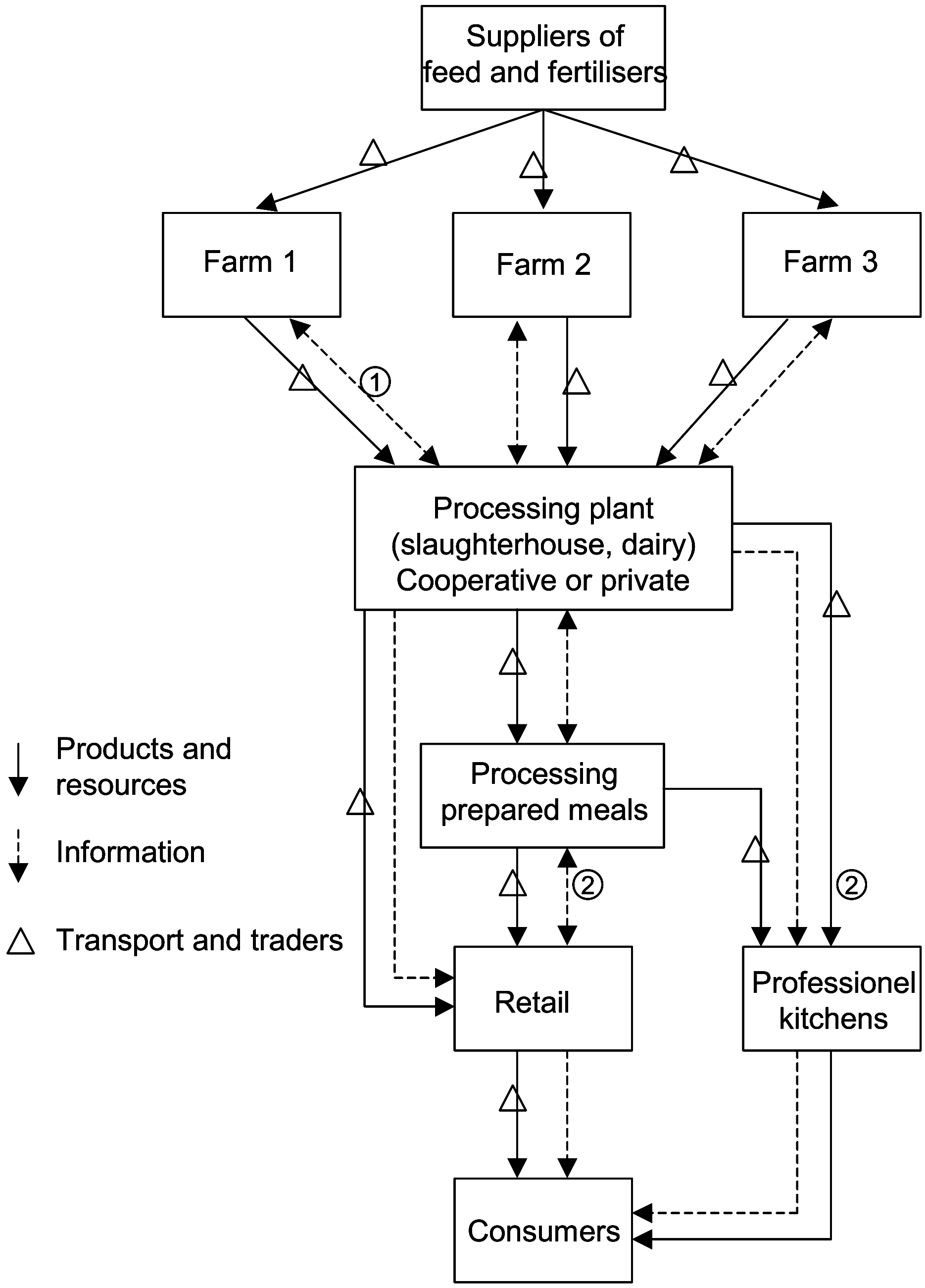
The quantitative, results based environmental information also has the advantage that they may describe the environmental impacts accumulated through the production chain, such as energy use and nutrient losses per kg product using LCA methods (as demonstrated in several papers in this volume). This would facilitate a product based environmental appraisal of food products in line with ideas of IPP (Anonymous, 2003). Therefore, it should be recommended to use quantitative environmental information based on the actual results from the production processes as part of QAS in the future. The existing LCA methodologies could be used as a starting point for this merge between IPP and QAS and supplemented with processes to include environmental aspects in product development (Nielsen & Wenzel, 2002).

To our knowledge the Dutch system "MPS" (Anonymous, 2004b) for green house production of flowers is the only environmental QAS for agricultural products which is both accredited by the retail sector in Europe, and builds on quantified information in a way that would fit in a product oriented environmental appraisal building on IPP/LCA principles. The MPS is accredited under EurepGAP but unlike most other GAP approaches this system is based on a quantification of the actual use of energy, pesticides and fertiliser per batch of flowers finished by the certified growers. Thus MPS is an example of how environmental QAS may be used for both documentation towards retail chains and benchmarking between growers. Some of the major retail chains have recently demanded MPS certification of flowerpots for their stores. The growers under MPS have to report their use of energy, pesticides and fertiliser every four weeks to the MPS organisation, where quarterly reports are compiled and sent out to growers with comparisons of their performance against standard limits and results of other growers. If a grower uses more of one input factor than the standard different measures are taken – depending on the severity of the exceeding and ultimately a grower may loose the certification for a period of time. The MPS approach demonstrates the feasibility of benchmarking between producers but the pot flower enterprises have a relatively short production chain from grower to retail compared with e.g. livestock products. However when looking at the existing dataflow on product quality in the pork sector including environmental data does not seem impossible from a practical viewpoint.

Fig. 1 gives an overview of the idea of creating an information flow on environmental profiles of livestock products along the physical flow of agricultural products from primary producers to retail. This idea is presently being tested in a case study involving a private Danish slaughterhouse and a number of its major suppliers of fattening pigs. The involved farms will establish Life cycle based green accounts of their production of fattening pigs with the help of local production advisors. These will be collected at the slaughter house for two purposes: 1.The accounts from different farms will in anonymous form be compared and fed back to farmers in a benchmarking exercise, where each farmer may assess his environmental performance in comparison with other producers delivering to this specific company. 2. The information on resource use and emissions in the primary production will be supplemented with environmental information from the slaughtering and other processes including transport to give an environmental profile per kg product delivered to the retailers and professional kitchens. It will be part of the project to explore which type of information the professional buyers will be interested in and how to convert LCA type of information into a format that is understandable for these stakeholders. It is not the idea to present this information to ordinary consumers because the LCA based information itself is assumed to be too complicated for laymen to relate to in the purchasing situation.

Conlusions

Rule based environmental quality assurance based on GAP is becoming part of the overall QAS in the food sector. The GAP approach may give some quality assurance for the food retail sector helping to reduce risks of food scandals from e.g. pesticide misuse analogously to HACCP systems reducing other food hazards. But the GAP approach found in most environmental QAS presently does not satisfactorily quantify the actual environmental performance on the farms nor does it allow benchmarking between farms, supplier cooperatives and products. Moreover, the GAP approach is not suitable for a product-oriented appraisal of environmental impacts from food products in line with the increasing interest in Integrated Product Policy. There is a need for development of an environmental quality assurance scheme that records and exchange information up- and downstream in the food chain and allows both primary producers and the food industry to continuously benchmark their performance and the retail sector to assure their customers that products are environmentally sustainable. Life Cycle Assessment and Life Cycle Management seems obvious tools for this and will be used in an attempt to develop such a system within a Danish slaughterhouse company.



1. Input/output accounts on farm level transformed into environmental profile per kg product from farms. Data for comparisons between farms re. ressource use and emissions per kg product, (benchmarking).

2. Documentation of environmental profiles of products accumulated through the chain in life cycle terminology

Figure 1. Simplified description of a food chain with physical flow of products and (potential) flow of information regarding quality and enviromental characteristics.

References

Anonymous, 1993. Assured produce scheme, available on-line (sept. 2004) at: *http://www.assuredproduce.co.uk/aproduce/*

Anonymous, 1996. Miljøbelastningen ved familiens aktiviteter (Environmental impact from the family's activities), Danish Consumer Board, report 1996. 1, 78 pp.

Anonymous, 2002. Evaluation of Environmental Product Declaration Schemes. Final Report. European Commission, DG Environment, 120 pp.

Anonymous, 2003. Integrated Product Policy – Building on Environmental Life-Cycle Thinking. Communication from the commission to the council and the European Parliament. Commision of the European Communities, Brussels, 18.06.2003,. COM (2003) 302 final.

Anonymous, 2004a: Danish IP: Gartneribrugets Afsætningsudvalg (GAU) Available on-line (sept. 2004) http://www.dansk-ip.dk.

Anonymous, 2004b: Miljø Programma Sierteelt. Available on-line (sept. 2004): www.my-mps.com

Arfini, F., Mancini, M.C, 2003. British Retail Consortium (BRC) Standard: a New Challenge for Firms Involved in the Food Chain. Analysis of Economic and Managerial Aspects. (ed.) G. Schiefer, U. Rickert. Proceedings of the 82nd Seminar of the European Association of Agricultural Economists (EAAE), May 14-16, Bonn, Germany. Vol. A, 23-31.

Brom, F.W.A., 2000. Food, consumer concerns, and trust: Food ethics for a globalizing market. Journal of Agricultural and Environmental Ethics 12, 127-139.

Cederberg, C., 2004. Systems for assurance of farm production in Western Europe. In. Thodberg et al. Produktorienteret miljøindsats i landbrugssektoren - forudsætninger og fremsyn. Report no. 900. App. E, p. 185-204 (in English). Available on-line (Sept. 2004) at www.mst.dk/hompage (publications)

Cederberg, C., Mattsson, B., 2000. Life Cycle Assessment of Milk Production - A Comparison of Conventional and Organic Farming. J. Cleaner Prod. 8, 49-60.

CFSAN, 2004. Center for Food Safety & Applied Nutrition. Hazard Analysis and Critical Control Point. HACCP overview. Availble on-line (Sept. 2004): http://vm.cfsan.fda.gov/~lrd/haccp.html

Dalgaard, T., Halberg, N. and Porter, J.R., 2001. A model for fossil energy use in Danish agriculture used to compare organic and conventional farming. *Agriculture, Ecosystems & Environment* **87(1):** 51-65.

Danske Slagterier, 2003: Description of the 'Danish Quality Guarantee', HACCP and traceability. Available on-line (Sept. 2004): http://www.danskeslagterier.dk/smcms/Danish_English/Danish_Quality/Index.htm?ID=1968

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

Delgado, C., M. Rosegrant, H. Steinfeld, S. Ehui, and C. Courbois, 1999. Livestock to 2020: The next food revolution. IFPRI. Discussion paper 28, 70 pp. Available on-line (August 2004): http://www.ifpri.org/2020/dp/dp28.pdf

EEA, 2001. Environmental benchmarking for local authorities: From concept to practice. Environmental issue report no. 20. European Environmental Agency.

Erzinger, S. and Badertscher Fawaz, R., 2001. Life Cycle Assessment of animal housing systems as part of an overall assessment. Proceedings International Conference on LCA in Foods, p. 118-122. SIK-Report Nr. 643. Swedish Institute for Food and Biotechnology, Gothenburg.

EurepGAP, 2004: FoodPLUS principles and generic rules for GAP, available on-line (sept. 2004) at: *http://www.eurep.org/sites/index_e.html*

Gerbens-Leenes, P.W., Nonhebel, S., 2002. Analysis – Consumption patterns and their effects on land required for food. Ecological Economics 42, 185-199.

Grønborg Eskesen, 1999. Danske gartnerier og MPS. Gartneritidende uge 29-30, 2003. Tilgængelig online: http://www.gartneritidende.dk/artikler.

Haas, G., Wetterich, F., Geier, U., 2000. Life Cycle Assessment Framework in Agriculture on the Farm level. Int. J. LCA 5, 345-348.

Halberg, N., 1999. Indicators of ressource use and environmental impact for use in a decision aid for Danish Livestock Farmers. Agriculture, Ecosystems and Environment 76, P. 17-30.

Halberg, N., Verschuur, G. & Goodlass, G. 2004. Farm level environmental indicators; are they useful? An overview of green accounting systems for European farms. Agriculture, Ecosystems and Environment. In Press.

Krieger, S., Schiefer, G., 2003. Quality Management Schemes in Europe and Beyond. In: G. Schiefer, U. Rickert (eds.). Proceedings of the 82nd Seminar of the European Association of Agricultural Economists (EAAE), May 14-16, Bonn, Germany. Vol. A, 35-47.

Larsson, I., 2004. Possible of the benefits from using LCA in the agro-food chain, example from Arla Foods. In: Halberg, N. (ed.) Life Cycle Assessment in the Agri-food sector. Proceedings of the 4th International Conference, October 6-8, 2003, Horsens, Denmark.

McKeown, P.J., 2001. The application of LCA methodology to the life of frozen vegetables. In: Proceedings of International Conference on LCA in Foods, Gothenburg, Sweden, 26-27 April 2001. SIK-document 143, 83-87.

Nienhoff, H.J., 2004. QS Quality and Safety: A Netchain Quality management Aggroach. In: G. Schiefer, U. Rickert (eds.). Proceedings of the 82nd Seminar of the European Association of Agricultural Economists (EAAE), May 14-16, Bonn, Germany. Vol. A, 627-629.

Nilsson, H., Tuncer, B, Thidell, Å., 2004. The use of eco-labeling like initiatives on food products to promote quality assurance – is there enough credibility? Journal of Cleaner production 12, 517-526.

Rosing, L. and Nielsen, A.M., 2004. When a hole matters - the story of the hole in a bread for French hotdog. In: Halberg, N. (ed.) Life Cycle Assessment in the Agri-food sector. Proceedings of the 4th International Conference, October 6-8, 2003, Horsens, Denmark.

Schiefer, G., 2004. From Enterprise Activity 'Quality Management' to Sector Initiative 'Quality Assurance': Development, Situation and Perspectives. In: G. Schiefer, U. Rickert (eds.). Proceedings of the 82nd Seminar of the European Association of Agricultural Economists (EAAE), May 14-16, Bonn, Germany. Vol. A, 3-20.

Spangenberg, J.H. & Lorek, S., 2002. Environmentally sustainable household consumption: From aggregate environmental pressures to priority fields of action. Ecological Economics, 43, 127-140.

Steenkamp, J-B. E.M., 1990. Conceptual Model of the Quality Perception Process. Journal of Business Research 21, 309-333.

Steenkamp, J-B. E.M., 1997. Dynamics in consumer behavior with respect to agricultural and food products. In: B. Wierenga, A. Van Tilburg, K. Grunert, J-B. E.M. Steenkamp, M. Wedel. (eds.). Agricultural Marketing and Consumer Behavior in a Changing World. Chapter 8, 143-188.

Swedish Seal, 2004. About Swedish Seal of Quality. Available on-line (Sept. 2004): www.svensktsigill.com.

Van Trijp, H.C.M., Steenkamp, J-B. E.M., Candel, M.J.J.M., 1997. Quality labelling as instrument to create product equity: The case of IKB in the Netherlands. In: B. Wierenga, A. Van Tilburg, K. Grunert, J-B. E.M. Steenkamp, M. Wedel (eds.). Agricultural Marketing and Consumer Behavior in a Changing World, 201-216.

Verbeke, W.A.J., Viaene, J., 2000. Ethical challenges for livestock production: Meeting consumer concerns about meat safety and animal welfare. Journal of Agricultural and Environmental Ethics 12, 141-151.

Wilting, H.C., Moll, H.C., Nonhebel, S., 1999. Direct and indirect energy use of European households. Groningen: IVEM, University of Groningen. IVEM report.

Sustainability in the Agrofood Sector

Klaas Jan Kramer & Marieke Meeusen
Agricultural Economic Research Institute (LEI), PO Box 29703, 2502 LS The Hague, the Netherlands
Phone: +31-70-3358245, Fax: +31-70-3615624, E-mail: klaasjan.Kramer@wur.nl; marieke.meeusen@wur.nl

Abstract
Sustainability is a hot issue. Society is demanding products and services that are produced and delivered in a sustainable manner. However, the concept of sustainability is very broad and needs concretising. To determine sustainability in agricultural product chains, LEI has developed a sustainability checklist. Elements of the checklist were used in the development of an annual social & environmental report on the Dutch horticultural sector. The availability of information/data and their application as indicators are a main factor in the determination of sustainability aspects. Analyses of chain information systems showed that these systems provide several linkages which are helpful in determining sustainability

Keywords: sustainability, checklist, indicators, information.

Introduction

Climate change, biodiversity, economic growth, human rights, poverty, labour conditions, etc. – all these factors are considered important regarding the sustainable development of our society, where such development means fulfilling the needs of the present generation without endangering the fulfilment of the needs of future generations. Companies are entering a new phase in which integrated responsibilities for mankind, the economy and the environment are becoming a prerequisite for good entrepreneurship. In other words, they are approaching sustainability from the perspective of people, planet and profit (the three P's).

The sustainability issue is increasingly under scrutiny at the chain level. The Dutch government is encouraging the agrofood business sector to take innovative steps towards sustainable development from the chain point of view. To gauge the sustainability of chains, the Agricultural Economics Research Institute (LEI) is involved in several research projects to determine and measure the progress of sustainability in agrofood chains. This paper presents some of the latest results of our sustainability chain research.

Motive

Indicators are a useful instrument to determine the level of a product's or a company's sustainability. Although there is still no precise definition of sustainable development, it is possible to begin working on indicators. The first question to ask is which issues and topics are involved in sustainable agrofood chains. The next question is how the sustainability of chains can be determined in a relatively straightforward manner. A set of simple instruments would be a helpful tool in this. These tools should preferably contain information that is already available. As a result, insight will be obtained into the sustainability of agrofood chains relatively quickly and without incurring excessive costs.

LEI was asked by the Dutch Ministry of Agriculture to draw up a checklist of issues or indicators to determine the sustainability of agrofood chains. LEI was also asked to provide an insight into the availability of the data used to determine sustainability. In this article we summarise the findings of several LEI projects aimed at determining sustainability with the help of chain information systems.

Aim

The aim of this paper is to provide greater insight into the possible ways of measuring the sustainability of agrofood chains. There are two distinct steps in this, viz. to establish:

- which indicators, topics and items are involved in sustainable agrofood chains, and
- what information is already available in order to make a relatively quick and cheap measurement.

Checklist for the sustainability of agrofood chains

Instruments that address the comprehensive scope of the definition of sustainability were expressly requested. These instruments were to elaborate on the three P's, viz. people (the socio-ethical component), planet (the environmental component) and profit (the economic component). In addition, they were to focus on the aggregate nature of the industry with respect to sustainability. They also had to take account of the consumer stage and other production chains. A fourth request was to respond to the challenge of harmonising potential within the business community. The instruments were to provide added value to current practice and to stimulate enterprises to become more aware, but they were not to set unreachable goals. These requests were the starting point for the checklist that was developed. This checklist provides an overview of all the categories and items that jointly determine the sustainability of agrofood chains.

An inventory of existing initiatives showed that although there are a great many of them they (a) have very little connection with each other, (b) generally have no bearing on the analysis level of chains, (c) have a specific bearing on the agricultural sector in only a limited number of cases, (d) pay little attention to the mutual trade-off between the various sustainability aspects, and (e) lack any conceptual or theoretical basis. An exception to this is the Sustainability Score Card, in which there is also an important connection with the private sector. However, the Score Card instrument is not made available to third parties. The Sustainable Corporate Performance model developed by the University of Groningen is also worthy of mention, but in 2002 no significant efforts had been made within this model to define indicators. The Life Cycle Analysis (LCA) was selected as one of the few instruments considered suitable for measuring environmental aspects along the whole chain.

In short, no system of measurable, scientifically-based indicators suitable for determining sustainability – either in the broadest sense (i.e. covering the three P's) or more narrowly – had been developed for agrofood chains. It was also apparent that the private sector was not ready for the far-reaching operationalisation of the concept. Although many companies are devoting attention to the concept of sustainability, they do not have a detailed operational plan for collecting periodic and standardised data. Therefore a checklist format was chosen, to be based on an inventory of many initiatives. The following aspects were examined: (a) the chosen system and set-up of measuring instruments, and (b) the chosen topics and the underlying issues.

The first version of the checklist for the measurement of sustainability in agrofood chains contained not only the outcomes and all topics but also a section on 'vision/mission/strategy' and one on 'measures'. This was intended to enable the measurement of the efforts and intentions behind sustainability. The third section comprised the 'outcomes'. Table 1 – which is a part of the AKK/LEI sustainability checklist (Meeusen & ten Pierick, 2002) – presents some of the results of the top-down approach.

In 2003, LEI began work on the second version of the checklist. A good basis was sought to serve as a conceptual framework for its development. This basis was found in Wood's Corporate Social Performance model (1991). LEI is now using this to draw up the new checklist. Here, links are made chiefly with the work of the Global Reporting Initiative (GRI).

Application: annual social & environmental report on the Dutch horticultural sector

LEI developed an annual social & environmental report covering 2002 for the Department of Greenhouse Horticulture of the Dutch Agricultural and Horticultural Organisation. To decrease the environmental impact of the Dutch greenhouse horticulture sector, the government and the sector drew up the voluntary agreement 'Greenhouse horticulture and the environment'. This agreement contains ambitious objectives for 2010 concerning increasing energy efficiency and decreasing the use of minerals and pesticides.

In 2000, employers' organisations and trade unions signed a declaration concerning the development of a voluntary agreement in the field of labour conditions. This agreement aims at decreasing the physical impacts (the major cause of absence due to illness and disablement) and improving the process of reintegration.

Table 1. Part of the LEI/AKK sustainability checklist (Meeusen & ten Pierick, 2002).

Dimension	Category	Aspect	Explanation
Planet	Transportation	Reducing freight transport	
	Energy	Reducing energy use	
		Renewable energy	Promoting its use
	Materials	Reuse of materials	
	Water	Water quality	Reducing emissions
	Air	Air quality	Reducing emissions
	Fauna	Biodiversity	Preventing the reduction in diversity of sorts and types of animals
Profit	Costs and efficiency	Price/quality ratio	Increasing the price/quality ratio of products and services
	Ethics in business-to-business context	Control and certification	Checking whether demands have been met
	Employment	Quantity of employment	Increasing the number of jobs
People			
	Working conditions	Workplace	Improving the location, interior (ergonomic) and safety
	Food safety		Reducing food-pollution components
	Norm and values	Emancipation	Stimulating integration of the elderly, handicapped, immigrants, women, etc.
	Social responsibility	Welfare	Contributing to the health, housing, safety, education, etc. of the community

The Dutch greenhouse horticultural sector has made some significant improvements in the field of sustainable production. The annual social & environmental report presents the steps taken by the sector to reduce its environmental impact and to become more socially sustainable. The 2002 social & environmental report partly covers some elements of the checklist and concerns the following topics: energy use, pesticides use, mineral use, environmental labelling, lighting, waste, labour, education, labour conditions and labour circumstances. These topics consist of various indicators. Figures 1 and 2 show the development of two different indicators: energy efficiency (energy per unit of production) and labour supply.

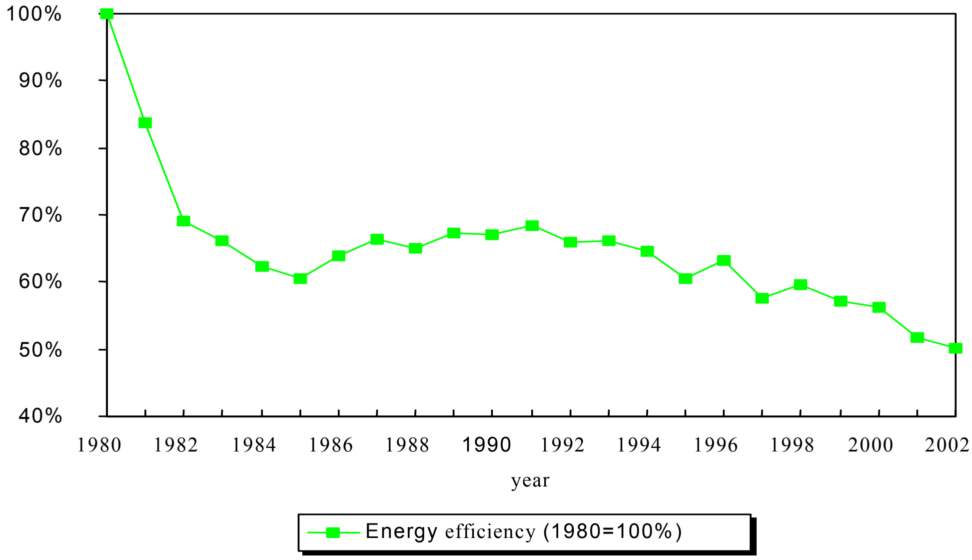


Figure 1. Development of the energy efficiency in the Dutch horticultural sector (van der Knijf & Benninga, 2003).

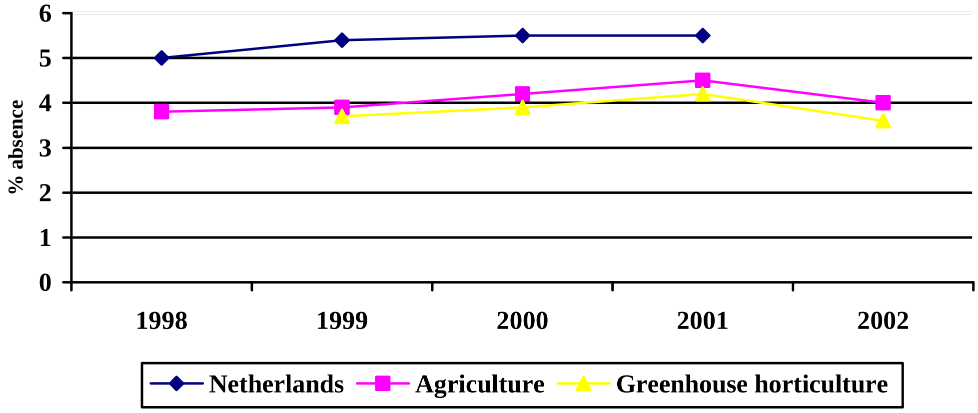


Figure 2. Development of absence due to illness (Netherlands Ministry of Social Affairs and Employment, 2002).

Information available for determining sustainability

This section 1) presents the results of research into the possibilities to use data from the Farm Accountancy Data Network (FADN) in determining sustainability, and 2) describes research into the possibilities to combine sustainability and chain information systems. In the latter project, the focus was on how chain information systems can be used to determine sustainability.

The FADN

The next step in developing the checklist was to formulate indicators. Based on several selection criteria, the appropriate indicators for sustainable agricultural chains were formulated. Selection criteria for good indicators were, for example, scientific validity, communicational quality, availability of data, representativeness, reproducibility, etc. In a LEI project carried out after indicators for sustainability had been developed, several possible indicators were judged with the help of several selection criteria. For the aspects of data availability, the possible indicators were judged on the information contained in the Farm Accountancy Data Network (FADN). Various data from a random sample of agricultural and horticultural holdings are stored in the network. Most aspects related to the economic dimension of sustainability can be determined, sometimes with another calculation step, with the help of the information contained in the FADN.

Possible indicators in the economic dimension are, for instance, number of jobs, financial results, investments in capital, R&D, human capital, and certification. Possible indicators in the social dimension are, for instance, number of registered complaints, number of days of illness (%), employee education/courses, contribution to local economies, and nature conservation. The FADN also contains information that can be used to determine some of the environmental aspects of sustainability, such as the total use of energy, mineral leaching, toxic emissions, waste, and land use.

In short, the FADN provides a sound basis for linking data with indicators. However, the emphasis here lies chiefly on the primary production phase; the FADN contains less information with which to determine the social aspects of the sustainability of agricultural holdings or chains. A second point is that for most of these environmental aspects, the information in the FADN first has to be transformed in order to create information about the environmental sustainability (this topic is dealt with later in this paper). This transformation also takes place in an environmental Life Cycle Assessment (LCA). In an LCA study of a product or service, flows within the economic system and flows out of the system into the environment are determined (i.e. all extractions of resources from the environment and emissions to the environment), when possible in a quantitative way. Based on this data, the potential impacts on natural resources, the environment and human health are assessed. The FADN serves as an input to carry out LCAs (Vrolijk et al., 2002).

Chain information

The availability of data is one of the impediments in the determination of sustainability in agro-food chains. In developing the annual social & environmental report, many data sources had to be consulted in order to obtain the appropriate information with which to determine the current level of sustainability. The collection of the necessary data is also the main focus of attention when carrying out an LCA. This factor led LEI to look at ways in which data can be supplied and collated in a different relationship. Together with ATO, LEI investigated the extent to which LCA and chain information systems (CISs) can be used to make a report assessing sustainability in the chain.

A CIS is defined as the way the chain actors share information. Two different systems can be distinguished: a system in which information is transferred by the various actors in the chain, and one which is placed outside the chain and acts like a type of database. In the latter system, all the actors in the chain have access to the system. Today, most CISs are developed to trace and/or track products. Tracking provides information about the current position in a chain, while tracing provides information about where the products have been and under what circumstances. Food safety and legislation are important motivations behind the growing attention to tracing and tracking and to CISs. Currently, the most important reason for a company to pay attention to tracing and tracking is to comply with laws and to limit damage claims. It is expected that companies will start to use tracing and tracking in a more positive way, viz. to better control the production processes (i.e. to shift from tracing to tracking).

In the research mentioned above, the possibilities for combining sustainability and CISs was investigated. This was carried out with the help of previous studies. Existing CISs were investigated in more detail: it was examined what kind of information is gathered in these systems and whether there are some links with sustainability.

Case study: the Groeinet information system (vegetables)
Groeinet develops registration programmes for environmental registrations, product registrations, logistics, etc. Agricultural companies can register their use of pesticides and fertilisers as well as the yield per crop. While the Groeinet information system was not developed to provide an insight into sustainability, the system does provide information useful in the determination of aspects of sustainability. For example, information about pesticides and fertilisers can be used to determine contributions to several environmental impacts. The production of these inputs is accompanied by, for example, the use of energy, which can be very high; for example, the fertiliser calcium ammonium nitrate (CAN) is produced from natural gas in the Netherlands, and it takes 0.7 m^3 of natural gas to produce 1 kg of CAN. However, databases with information about the energy contents of inputs are necessary to determine the total energy use. The use of pesticides leads to emissions of toxic compounds to water, soil and air. With information about the eco-toxic effects of pesticides use, the recorded information regarding pesticides can be used to determine sustainability as regards the impact on the quality of water, soil and air.

The case studies showed that the Groeinet information system contains information relevant to sustainability. However, additional information is required to translate the used inputs into useful information about sustainability. For some aspects of sustainability this can be done easily. The case studies showed that this is the case for environmental sustainability aspects. However, the determination of aspects of social sustainability is more difficult. This is because CISs focus on products. Environmental aspects could also be related to products, as in an LCA. Aspects of social and economic sustainability are more related to processes than to products. However, there are possibilities to relate social aspects to a product, for example, the number of labour hours or the labour happiness level required to produce a certain number of products. Additional research is required to work this out in further detail (Kramer et al., 2002).

Summary

This article focused on the development of issues and themes to determine the progress in making agricultural chains sustainable. For some sustainability aspects, no indicators are available. However, for the Dutch horticultural sector, some environmental and social sustainability indicators have been developed. To determine environmental sustainability in chains, elements of LCA can be used. The development of an annual social & environmental report on the Dutch horticultural sector showed that the availability of information is still a major impediment to the determination of sustainability. This problem occurs mainly at the level of the production chain. On the farm level, several systems (mainly the FADN) contain the necessary information. A possible solution to this would be to combine chain information systems with sustainability.

References

Netherlands Ministry of Social Affairs and Employment (2002). Nulmeting en onderzoek stand der techniek fysieke en psychische belasting arboconvenant agrarische sectoren, The Hague, the Netherlands.

Meeusen, M.J.G. and E. ten Pierick (2002), Meten van duurzaamheid

Naar een instrument voor agroketens, Landbouw-Economisch Instituut (LEI), The Hague, the Netherlands

Vrolijk, H.C.J., G. Cotteleer, K.J. Kramer, T.C. van Leeuwen and H.H. Luesink (2003). Performance indicators. Landbouw-Economisch Instituut (LEI), The Hague, the Netherlands

Kramer, K.J, Thors, M. and J. Wolfert. (2003). Duurzaamheid in agrofood ketens, Landbouw-Economisch Instituut (LEI), The Hague, the Netherlands

Knijff, van der A. and J. Benninga (2003). Energie in de glastuinbouw van Nederland; ontwikkelingen in de sector en op de bedrijven t/m 2002. Landbouw-Economisch Instituut (LEI), The Hague, the Netherlands.

Wood, D.J. (1991). 'Corporate Social Performance Revisited'. Academy of Management Review, vol. 16, no. 4, 691-718.

Life Cycle-Based Sustainability Indicators for Assessment of the U.S. Food System

Keoleian, G.A. and. Heller, M.C.*

**Center for Sustainable Systems, School of Natural Resources and Environment, 430 East University, University of Michigan, Ann Arbor, MI 48109-1115 U.S.A.;*
phone: 1 734 764 3194; fax: 1 734 647 5841; gregak@umich.edu

Abstract

The overall sustainability of any food system is influenced not only by agricultural practices but also by food consumption behaviors, food processing and distribution activities. A product life cycle approach provides a useful framework for studying the links between production and consumption activities. The United States has a highly productive agricultural system as measured by the total amount of food output. Over the last century dramatic changes to this system have been implemented to enhance this productivity metric but the environmental, social, and economic consequences have also been significant. This paper highlights a select set of indicators covering the life cycle management of the entire food system. Indicators address economic, social, and environmental aspects of each life cycle stage: origin of (genetic) resource; agricultural growing and production; food processing, packaging and distribution; preparation and consumption; and end of life.

Introduction

A recent assessment of the sustainability of the U.S. food system was conducted by the Center for Sustainable Systems. A life cycle framework was used to identify and organize a set of social, economic and environmental indicators for evaluating the current U.S. food system. Even though a comprehensive life cycle assessment (LCA) of the food system is not currently possible, using a life cycle framework does provide a systematic basis for developing indicators. A matrix defined by life cycle stages and the three dimensions of sustainability served to group the indicators. A total of 57 indicators were proposed and evaluated. The full set of indicators are presented in two publications by the authors (Heller and Keoleian, 2003; Heller and Keoleian, 2000). The purpose of this paper is to highlight key findings of our investigation. In particular, current trends for a select set of ten indicators that threaten the long-term economic, social, and environmental sustainability of the US food system are discussed.

Method

Table 1 presents the full matrix of sustainability indicators developed in Heller and Keoleian 2003. The rows represent major stages of the food system; indicators for each stage are categorized into the “triad” of sustainability: economic, social, and environmental. In many incidences, this division is somewhat arbitrary since particular indicators often address more than one aspect of sustainability. Also identified in Table 1 are the primary stakeholders involved or influential in each stage of the food system.

Table 1. Life Cycle Sustainability Indicators for the Food System (Heller and Keoleian 2003).

Life cycle stage	**Stakeholders**	**INDICATORS**		
		Economic	Social	Environmental
Origin of (genetic) resource – seed production, animal breeding	Farmers Breeders Seed Companies	-degree of farmer/operator control of seed production/breeding	-diversity in seed purchasing and seed collecting options -degree of cross-species manipulation	-ratio of naturally pollinated plants to genetically modified/ hybrid plants per acre -reproductive ability of plant or animal -% of disease resistant organisms
Agricultural growing and production	Farm operators Farm workers Ag. Industry Ag. Schools Government Animals	-rates of agricultural land conversion -output/input productivity -% return on investment -cost of entry to business -farmer savings and insurance plans -flexibility in bank loan requirements to foster environmentally sustainable practices -level of gov't support	-average age of farmers -diversity and structure of industry, size of farms, # farms per capita -hours of labor/ yield and / income -avg. farm wages vs. other professions -# of legal laborers on farms, ratio of migrant workers to local laborers, -% workers with health benefits. -# of active agrarian community organizations -% of ag. Schools that offer sustainable ag. programs, encourage sustainable practices -# animals/unit, time animals spend outdoors (animal welfare)	-rate of soil loss vs. regeneration -soil microbial activity, balance of nutrients/acre -quantity of chemical inputs/ unit of production -air pollutants/ unit of production -number of species/acre -water withdrawal vs. recharge rates -# of comtaminated or eutrophic bodies of surface water or groundwater -% waste utilized as a resource -veterinary costs -energy input/ unit of production -ratio of renewable to non-renewable energy -portion of harvest lost due to pests, diseases
Food processing, packaging and distribution	Food processors Packaging providers Wholesalers Retailers	-relative profits received by farmer vs. processor vs. retailer -geographic proximity of grower, processor, packager, retailer	-quality of life and worker satisfaction in food processing industry -nutritional value of food product -food safety	-energy requirement for processing, packaging and transportation -waste produced/ unit of food -% of waste and byproducts utilized in food processing industry -% of food lost due to spoilage/mishandling
Preparation and consumption	Consumers Food service Nutritionists/ Health professionals	-portion of consumer disposable income spent on food -% of food dollar spent outside the home	-rates of malnutrition -rates of obesity -health costs from diet related disease/conditions -balance of average diet -% of products with consumer labels -degree of consumer literacy regarding food system consequences, product quality vs. appearance, etc. -time for food preparation	-energy use in preparation, storage, refrigeration -packaging waste/ calories consumed -ratio of local vs. non-local and seasonal vs. non-seasonal consumption
End of life	Consumers Waste managers Food recovery & gleaning orgs	-ratio of food wasted to food consumed in the US -$ spent on food disposal	-ratio of (edible) food wasted vs. donated to food gatherers	-amount of food waste composted vs. sent to landfill/incinerator/ waste water treatment

Table 2 summarizes the scope and boundary conditions for the assessment. The current agricultural system is highly productive, yielding an estimated 161 million metric tons of edible food in 1995 for a U.S. population of 263 million. As a result of affordable food and food assistance, more than 90% of U.S. households were food secure, meaning they had assured access at all times to enough food for an active healthy life. Still, 9.7% of U.S. households – about 10 million – were food insecure over the 1996-1998 period. The food energy available for consumption, based mainly on national disappearance of food was 15900 kJ (3800 kilocalories) per capita per day in 1994 (NASS 1999). However, this value includes food that is wasted at the retail and consumer level. According to USDA's Continuing Survey of Food Intakes by Individuals (CSFII), the surveyed caloric intake was 8382 kJ based on 1994-1996 surveys (Tippett and Cleveland 1999).

Table 2. Scope and boundary condition for this sustainability assessment.

• System boundary encompass: origin of (genetic) resource; agricultural growing and production; food processing, packaging and distribution; preparation and consumption; and end of life management • Disaggregation of food from other agricultural production is not completely possible • U.S. exports and imports of agricultural products are both significant but indicators are not corrected for agricultural trade ratios • Analysis is conducted at a national scale recognizing, however, that there is significant regional variation in indicators

Key Findings

A select set of indicators from Table 1 is presented here. These were chosen by the authors based on their current importance in influencing the long-term sustainability of the US food system.

1. Rapid conversion of prime farmland

The land available for agriculture is an primary indicator of its stability. According to the National Resources Conservation Service, cropland and pastureland (including CRP land) totaled 213.4 million ha in 1997, a 5% decrease from 1982 (NRCS 1999). 5.7 million ha of cropland and pastureland were converted to urban or built-up land over this 15 year period, amounting to over 50% of the decrease. As prime farmland is being developed, less stable non-prime farmland in arid regions is being added to the base, leading to increased erosion rates and irrigation demands (Harlin 1995).

2. Depletion of topsoil exceeds regeneration

The National Resources Inventory of the USDA (1999b) reports that 1700 megatonnes (Mt, = million metric tonnes) of soil eroded from U.S. land in 1997, 760 Mt from wind and 960 Mt from sheet and rill (caused by water) (NRCS 1999). Improvements have been made in recent years: the number reported for 1982 is 2780 Mt (NRCS 1999). 45.3 million ha (30% of cropland) were determined to be excessively eroding in 1997 (erosion rates greater than that deemed tolerable), totaling to 1200 Mt of eroded soil per year (NRCS 1999). If the 1700 Mt of topsoil lost in 1997 were evenly distributed over all of the U.S. cropland, the average rate of erosion would be 9.9 tonnes per ha per year. This translates into 2.5 cm of topsoil lost from all U.S. cropland every 34 years. It should be quickly recognized that this practice is not sustainable.

3. Rate of groundwater withdrawal exceeds recharge in major agricultural regions

Agriculture greatly affects the quantity of water consumed in the U.S., primarily through irrigation of crops and through livestock production. In 1995, 507 million m^3 per day (134,000 million gallons per day) of freshwater were withdrawn for irrigation purposes (39% of total freshwater withdrawal), 185 million m^3 per day of this from groundwater sources. Water consumption for livestock totaled 20.8 million m^3 per day in 1995, 41% of which was from groundwater (Solley et al. 1998). The concern is that, in certain regions of the country, withdrawal from groundwater sources is exceeding the natural recharge rate of aquifers. An excellent case-in-point is the Ogallala aquifer in the High Plains states. The Ogallala aquifer is largely a nonrenewable resource that has been mined at rates that greatly exceed recharge. Pumping from the aquifer is measured in feet per year while replacement, trickling in from the surface only, occurs at less than an inch a year (Opie 1993). Irrigation rates have decreased slowly over recent years, but current withdrawal rates still cannot be sustained long into the future.

4. Losses to pests increasing

According to the Census of Agriculture, \$7600 million were spent on agricultural chemicals in 1997, up from \$4700 million in 1987 (USDA 1999a). Average application rates appear to have decreased, however, from a two decade high of 2.0 kg ha^{-1} in 1987 to 1.3 kg ha^{-1} in 1996 (Kellogg et al. 1999). Interestingly, while pesticide use is generally seen as profitable in terms of direct crop returns, it has not necessarily led to decreases in crop loss. Even with a tenfold increase in insecticide use from 1945 to 1989, total crop losses from insect damage have nearly doubled from 7% to 13% (Pimentel et al. 1991). This rise in crop losses is partly caused by changes in agricultural practices such as abandoning crop rotations and increased crop homogeneity.

5. Reduction in genetic diversity

Modern agricultural systems have also become increasingly genetically uniform. Only 10-20 crops provide 80-90% of the world's calories (Brown 1981). In the U.S., 42% of the soybeans, 43% of the corn, and 38% of the wheat grown in 1980 were dominated by the top 6 varieties (Duvick 1984). Often, these varieties originate from an even smaller genetic base. For example, the hundreds of corn hybrids grown in the U.S. are largely based on about 12 inbred lines that originated from a few open-pollinated varieties of a single race of the some 200 known races of corn. (Soule et al. 1990). Repeated warnings have been sounded from the research community about the extreme vulnerability associated with this limited genetic diversity (CAST 1999; Wood and Lenne 1999). Lack of biodiversity in crops leads to pest and disease susceptibility (National Research Council 1972; Brown 1983; Altieri 1994). Insufficient crop genetic diversity led to the corn leaf blight in southern U.S. in 1970 (CAST 1999) and is partly responsible for outbreaks of *Fusarium* head blight in wheat and barley in Minnesota and the Dakotas (McMullan et al. 1997). The forces driving genetic uniformity also lead to abandonment and loss of locally adapted varieties that are the necessary resources to meeting future plant breeding challenges.

6. Poor economic conditions for farmers

Industrialization of agriculture has made production extremely capital intensive, leading to high cost of entry. An estimated $500,000 in assets is needed to support a farm household (ERS 2000). This, combined with return on investment considerably lower than can be received in other business ventures, is contributing to declining numbers of young farmers. Rates of return on farm business equity in 1998 were reported at 2.15%, and have averaged –0.3% since 1980 (3.8% since 1990) (ERS 2000). Nearly half (48%) of all farms reported a negative net cash return (net loss) in 1997. Ninety-two percent of the farms reporting losses in 1997 were relatively small, with sales worth less than $50,000 (USDA 1999a). Still, farm households in the U.S. receive incomes on par with average U.S. households. The 1997 average household income for farm operator households was $52,300. However, 89% of this household income comes from off-farm sources (USDA 1999b).

Providing inexpensive food for Americans has long been a central tenet in U.S. agricultural policy. Yet, as food processing, handling, and marketing have increased, the farmer has received smaller and smaller portions of the American food bill. USDA estimates that the farmer's gross return on a consumer's dollar spent on food in 1998 was 20 cents (Elitzak 1999a) (in 1975 it was 40 cents (Elitzak 1999b)). The remaining 80% of the food bill is distributed among marketing labor, packaging, advertising and other categories.

7. Declining entry of young farmers into the profession

Another prominent trend on America's farms is the advancing age of farm operators. According to the Census of Agriculture, the average age of farm operators in 1997 was 54.3 years, and 61% of the operators were 55 and over. In 1954, only 37% of farmers were 55 and over. Comparatively, 11.7% of the civilian labor force was age 55 or above in 1997 (18% in 1954) (ERS 2000). The percentage of farmers under 35 years of age dropped from 15% in 1954 to 7.8% in 1997 (ERS 2000). Furthermore, farm labor has dropped significantly over the past 50 years, from 9.9 million workers in 1950 to 2.8 million in 1998 (USDA 1999b).

8. Obesity rates are rising along with the costs of diet related diseases

Data from the National Health and Nutrition Examination Surveys of 1977-80 and 1988-1994 demonstrate that the prevalence of obesity is on the rise throughout the American population. The number of overweight individuals rose over the time between surveys from 25.4% to 34.9% among American adults, from 7.6% to 13.7% among children ages 6-11 years, and from 5.7% to 11.5% among adolescents (Nestle and Jacobson 2000). Under an updated definition presented in the 2000 Dietary Guidelines (USDA 2000), 60% of males and 46% of females 20 years and over were overweight in 1994-1996 (Tippett and Cleveland 1999). Increasingly, scientific studies confirm that America's diet of high fat intakes and low intakes of whole, fiber-containing foods such as whole grains, vegetables, and fruits has a significant impact on our health, quality of life, and longevity. Diet is a significant factor in the risk of coronary heart disease, certain types of cancer, and stroke – the three leading causes of death in the United States (National Research Council 1989). Estimates of diet-related medical costs, loss of productivity, and value of premature deaths reach $71,000 million per year (Frazão 1999). Estimates of the direct health care costs of obesity alone range from $39,000 million (Allison 1999) to $52,000 million (Wolf and Colditz 1998) annually. The prevalence of overweight and obese Americans was highlighted as a major agenda issue at the National Nutrition Summit in May of 2000 (Scannell et al., 2000).

9. A large fraction of edible food is wasted

41.2 Mt of food is wasted at home and at food service establishments, amounting to 26% of the edible food available for human consumption in the U.S. Fresh fruits and vegetables accounted for 19% of these losses, and an additional 18% was fluid milk (Kantor et al., 1997). Examinations of household garbage by researchers at the University of Arizona concluded that large quantities of single food items – entire heads of lettuce, half-eaten boxes of crackers – accounted for a larger share of household food loss than did plate scraps. They also found that specialty products such as sour cream, hot dog buns, or impulse items had a higher frequency in household garbage than did frequently purchased staples like bread, milk and cereal (Kantor et al., 1997).

10. Heavy reliance on fossil fuel

In total, providing the 15900 kJ of food energy available per capita per day in the United States is estimated to consume 10.8 million TJ annually. This represents about 10% of the total energy consumed in the United States (EIA 2001). By our estimates, therefore, it takes about 7.3 units of (primarily) fossil energy to produce one unit of food energy in the U.S. food system. This value is somewhat lower than estimates reported by others. Pimentel and Pimentel 1996 and Hall et al. 1986 both estimate 10 units of input energy per unit of output food energy.

Conclusions

Land, sufficient topsoil, water, and human capital are all essential inputs for a sustainable food system. A sustainable food system must also be founded on a sustainable diet. In the most general sense, this would be a diet that matched energy intake with energy expenditure while supplying necessary nutrients for a healthy lifestyle. The greatest leverage point for enhancing the sustainability of the U.S. food system lies with the level of consumption and amount of food waste. Significant improvements in diet not only have direct health benefits and reduced costs of diet related diseases but also more than proportionally reduce environmental impacts from agricultural production. The opportunity to reduce food production is tremendous by limiting excess consumption estimated at 8382 kJ per capita per day and edible food waste estimated at 26%. A reduction by one third is not unrealistic.

The economics of the U.S. food system also needs some fundamental adjustments to reverse unsustainable social and environmental impacts. Entry of young farmers into the profession is declining and production is shifting to larger scale farms, which are less ecologically sustainable. A systems-based solution would combine a reduction in food consumption and waste while maintaining revenues to farmers for less food output. The disposable income spent on food could be held constant in this scenario and costs of diet related diseases would be dramatically reduced. Until society places a higher value on food, the reported unsustainable patterns will continue. It is clear that governmental policies that address both production and consumption are necessary to advance the sustainability of our food system.

References

Allison, D. B., 1999. "The Direct Health Care Costs of Obesity in the United States." *American Journal of Public Health* **89**(8): 1194-1199.

Altieri, M. A., 1994. *Biodiversity and Pest Management in Agroecosystems*. New York, Haworth Press.

Brown, N. J., 1981. *Biological Diversity: The Global Challenge*. The U.S. Strategy Conference on Biological Diversity, U.S. Department of State Publication.

Brown, W. L., 1983. *Economic Botany* **37**: 1-12.

CAST, 1999. Benefits of Biodiversity, Council for Agricultural Science and Technology.

Duvick, D. N., 1984. "Genetic Deversity in Major Farm Crops on the Farm and in Reserve." *Economic Botany* **38**(2): 161-178.

EIA, 2001. *Annual Energy Review 2000*. Washington, DC, Energy Information Administration, U.S. Dept. of Energy.

Elitzak, H., 1999a. "Desire for Convenience Drives Marketing Costs." *Food Review* **22**(3): 23-25.

Elitzak, H., 1999b. Food Cost Review, 1950-97. Washington, DC, Economic Research Service, USDA.

ERS, 2000. The Aging of Farm Operators and Participation of Beginning Producers in Farming, Economic Research Service, USDA.

ERS, 2000. Farm Business Balance Sheet and Financial Ratios, Economic Research Service, USDA.

Frazão, E., 1999. High Costs of Poor Eating Patterns in the United States. *America's Eating Habits: Changes and Consequences.* E. Frazão. Washington, DC, Economic Research Service, USDA. Agriculture Information Bulletin **No. 750**.

Furst, T., M. Connors, Bisogni, C.A., Sobal, J., Falk, L.W., 1996. "Food Choice: A Conceptual Model of the Process." *Appetite* **26**(3): 247-265.

Hall, C. A. S., C. J. Cleveland, Kaufmann, R., 1986. *Energy and Resource Quality*. New York, Wiley Interscience.

Harlin, J. M., 1995. *Rangeland to Cropland Conversions in the Western States: Implications for Soil Loss and Sustainability*. Forum of the Association for Arid Land Studies XI, Lubbock, TX, International Center for Arid and Semiarid Land Studies.

Heller, M. C., and Keoleian, G. A., 2000. Life Cycle-based Sustainability Indicators for Assessment of the U.S. Food System. Center for Sustainable Systems, Ann Arbor, MI. CSS00-04, available: http://css.snre.umich.edu/css_doc/CSS_2000_4.pdf.

Heller, M.C., G.A. Keoleian (2003) Assessing the sustainability of the U.S. food system: a life cycle perspective. *Agricultural Systems* **76**: 1007-1041.

Kantor, L. S., K. Lipton, Manchester, A., Oliveira, V., 1997. "Estimating and Addressing America's Food Losses." *Food Review* **20**(1).

Kellogg, R. L., Nehring, R., Grube, A., Plotkin, S., Goss, D. W., and Wallace, S., 1999. Trends in the Potential for Environmental Risk from Pesticide Loss from Farm Fields. Proceedings of The State of North America's Private Land Conference (Chicago, IL, January 19-21, 1999), National Resources Conservation Service, USDA.

McMullan, M., R. Jones, Gallenberg, D., 1997. "Scab of wheat and barley: A re-emerging disease of devastating impact." *Plant Disease* **81**: 1340-1348.

NASS, 1999. Chapter XIII: Consumption and Family Living. *Agricultural Statistics 1999*. Washington, DC, National Agricultural Statistics Service, USDA.

National Research Council, 1972. *Genetic Vulnerability of Major Crops.* Washington, DC, National Academy Press.

National Research Council, 1989. *Diet and Health: Implications for Reducing Chronic Disease Risk*. Washington, DC, National Academy Press.

Nestle, M. and M. F. Jacobson, 2000. "Halting the Obesity Epidemic: A Public Health Policy Approach." *Public Health Reports* **115**(Jan/Feb.): 12-24.

NRCS, 1999. *1997 National Resources Inventory Summary Report*. Washington, DC, Natural Resources Conservation Service, USDA.

Opie, J., 1993. *Ogallala: Water for a dry land. Lincoln*, University of Nebraska Press.

Pimentel, D., L. McLaughin, Zepp, A., Lakitan, B., Kraus, T., Kleinman, P., Vancini, F., Roach, W. J., Graap, E., Keeton, W. S., and Selig, G., 1991. Environmental and economic impacts of reducing U.S. agricultural pesticide use. *Handbook on Pest Management in Agriculture*. D. Pimentel. Boca Raton, FL, CRC Press**:** 679-718.

Pimentel, D. and M. Pimentel, 1996. Energy Use in Fruit, Vegetable, and Forage Production. *Food, Energy, and Society*. D. Pimentel and M. Pimentel. Niwot, CO, University Press of Colorado**:** 131-147.

Scannell, K. M., M. E. Conway, Gordner, R. L., Klein, L. J., and Zorn, M, 2000. National Nutrition Summit: information resources, National Library of Medicine (U.S.).

Solley, W. B., R. R. Pierce, Perlman, H. A, 1998. *Estimated Use of Water in the United States in 1995*. Denver, CO, U.S. Department of the Interior.

Soule, J., D. Carre, Jackson, W., 1990. *Ecological Impact of Modern Agriculture. Agroecology*. C. R. Carroll, J. H. Vandermeer and P. M. Rosset. New York, McGraw-Hill Publishing Co.**:** 175.

Tippett, K. S. and L. E. Cleveland, 1999. How Current Diets Stack Up: Comparison with Dietary Guidelines. *America's Eating Habits: Changes and Consequences*. E. Frazão. Washington, DC, Economic Research Service, Food and Rural Economics Division, USDA. Agriculture Information Bulletin **No. 750**.

USDA, 1999a. *1997 Census of Agriculture*, U.S. Dept. of Agriculture.

USDA, 1999b. *Agriculture Fact Book 1999*. Washington, DC, U.S. Dept. of Agriculture, Office of Communications.

USDA, 2000. *Dietary Guidelines for Americans*. Washington, DC, U.S. Dept. of Agriculture.

Wolf, A. M. and G. A. Colditz, 1998. "Current estimates of the economic cost of obesity in the United States." *Obesity Research* **6**: 97-106.

Wood, D. and J. M. Lenne, Eds., 1999. *Agrobiodiversity: Characterization, Utilization, and Management.* New York, CABI Publishing.

Most significant substances of LCA to Mediterranean Greenhouse Horticulture

Antón, M.A.[(a)]; Castells, F.[(b)]; Montero, J.I. [(a)]; Muñoz, P.[(a)]*

[a] *Institut de Recerca i Tecnologia Agroalimentària. Centre de Cabrils. Ctra. de Cabrils s.n. 08348, Cabrils (Barcelona). Spain. Tel: 34937507511 Fax: 34937533954* assumpcio.anton@irta.es

[b] *Department of Chemical Enginering. ESTEQ. Universitat Rovira Virgili 43007 Tarragona. Spain*

Abstract

LCA has been used as a tool to identify the environmental burdens associated to protected cultivation in the Mediterranean area. A tomato crop was cultivated in a steel-framed greenhouse; plants were transplanted directly in the soil. With the exception of toxicity indicators, the main sources of environmental impact were production and use of the fertilizer needed, and the manufacture of the greenhouse structure. From the 470 substances considered in this study, only 21 contributed more than 5% of the total impact of one category. Among the 21 substances, 16 contributed to air or water emissions, being 6 of them pesticides, and 5 were non-renewable resources. Results from this study also suggest that there is a potential for simplifying LCA methodology by considering a restricted number of substances.

Keywords: simplified LCA, protected horticulture

Introduction

Previous works on the application of LCA in protected horticulture were focused in North European Regions. They showed that reducing the heating requirements must be a priority in order to limit the environmental load. (Jolliet, 1993, Bucher y col., 1996, Nienhuis y col., 1996, Jungbluth y col., 2000, Van Woerden, 2001). Regarding protected horticulture in southern areas, where there is no need of heating or these needs are much lower, there was a strong lack of information on which greenhouse production and handling processes have bigger effects on the environmental impact of greenhouse production. Antón (2004) studied the main environmental burdens associated to protected horticulture in Mediterranean countries comparing different production means. In this work we have evaluated the traditional soil cropping system and try to identify the main contributor substances in order to promote simplified LCA at more local level.

Material and methods

A tomato crop was cultivated in a traditional greenhouse with steel frame and PE film as the cover. Plants were transplanted directly in the soil. This study takes into consideration only the means of production. Therefore the analysis finishes at the moment that tomatoes are harvested, "farm's gate". The kg of tomato produced was chosen as the functional unit. Agronomic data were collected from different tomato growers of Maresme, an area located 30 km North from Barcelona.

In order to know the relative importance of the different processes, greenhouse structure, pesticides and fertiliser production, greenhouse, pest, fertirrigation and waste management were assessed as subsystems.

1. *Tomato production*
   Due to its complexity and in order to facilitate study, the tomato production was divided into five sub-systems:
   1.a) Greenhouse management during tomato production, GM
   1.b) Fertilizer production, F
   1.c) Fertilization and irrigation, FR
   1.d) Pesticide production, P
   1.e) Pest management, PM
2. *Manufacture*
   Two further subsystems were added to take into consideration the different process and materials used in the greenhouse structure and for the auxiliary equipment, the fertilization and irrigation system:
   2.a) Greenhouse structure, G
   2.b) Auxiliary equipment, R
3. *Waste*
   The final system analysed included the management of waste generated during and at the end of greenhouse crop cultivation, W

Results

Table 1 shows the overall impact of the global process of tomato greenhouse cropping for the different impact categories and their contribution to the different subsystems considered. With relation to climate change, it is important to point out the negative impact due to CO_2 fixation by the crop.

Fertiliser production was the main stage in the cycle that contributed to climate change (83%) and depletion of non-renewable resources (65%). During crop production, pesticides were mainly responsible for the toxicity indicator scores, while the fertirrigation was the main contributor to eutrophication (60%) mainly due to nitrogen compounds emissions. The greenhouse structure manufacturation (steel frame and cladding) mainly contributed to the categories of photochemical oxidant formation (45%) caused by hydrocarbons emissions to the air, climate change (34%), mostly CO_2 emissions, and air acidification (28%), principally due to release of NO_x and $SO_{x.}$

From the 470 substances considered in this study, only 21 contributed more than 5% of the total impact of one category. Among the 21 substances, 16 contributed to air or water emissions, being 6 of them pesticides, and 5 were non renewable resources (table 2). From those 10 emitted substances non pesticides, 8 were cited also as main contributor substances in Cowell (1998). These substances and their contribution of each category are shown in figure

1. Toxicity potentials are mainly due to the use of pesticides. Nevertheless it has to be mentioned that pesticide types and application may change from one exploitation to another as a function of the pests affecting the crop every season.

Conclusions

Further research must be orientated towards reducing the environmental impact of the materials used in the structure and looking forward a more rational management criteria in the supply of nutrients to the crop in order to reduce fertiliser use and avoid the leachates

Since only a reduced number of substances are the main contributors for the environmental impact, it may be possible to simplify the assessment by focusing on a restricted number of substances.

Acknowledgements

The authors thank Montse Martí and Jordi Ariño for providing useful information related to tomato crop farmers. This research was partially supported by INIA nº SC00-080-C2 and nº RTA03-096-C5-2.

Literature cited

Antón, A. (2004). *Utilización del Análisis del Ciclo de Vida en la Evaluación del Impacto Ambiental del cultivo bajo Invernadero Mediterráneo*. Tesis doctoral. Programa d'Enginyeria Ambiental. Universitat Politècnica de Catalunya. Barcelona.

Bucher, J., Chelkova, N., van der Ende, J.P., van der Hulst, P., Fernández Soler, M., Szarvas, G., Vrbensky, R. (1996). *Tomato North-South: Recommendations for the improvement of environmental quality of tomato production using different environmental labels*. Centre of Environmental Science, Leiden University. Leiden. xxxi+96+31

Cowell, S.J. (1998). *Environmental Life Cycle Assessment of Agricultural Systems: Integration Into Decision-Making*. Ph. D. thesis. Centre for Environmental Strategy. University of Surrey. Guildford.

Jolliet, O. (1993). Bilan écologique de la production de tomates en serre. *Revue S. Vitic. Arboric. Hortic.* 25 (4): 261-267.

Jungbluth, N., Tietje, O., Scholz, R.W. (2000). Food Purchases: Impacts from the Consumers' Point of View Investigated with a Modular LCA. *International Journal of Life Cycle Assessment.* 5: 134-142.

Nienhuis, J.K., Vreede, P.J.A.d., De Vreede, P.J.A., Brumfield, R.G. (1996). Utility of the environmental life cycle assessment method in horticulture. En: *Proceedings of the XIIIth International Symposium on Horticultural Economics, Rutgers*. (Ed). New Brunswick, New Jersey, USA. No. 429: 531-538.

Van Woerden, S. (2001). The application of Life Cycle Analysis in glasshouse horticulture. En: *International Conference LCA in Foods*. (Ed). Gothenburg. 136-140.

Antón, A. (2004). *Utilización del Análisis del Ciclo de Vida en la Evaluación del Impacto Ambiental del cultivo bajo Invernadero Mediterráneo*. Tesis doctoral. Programa d'Enginyeria Ambiental. Universitat Politècnica de Catalunya. Barcelona.

Bucher, J., Chelkova, N., van der Ende, J.P., van der Hulst, P., Fernández Soler, M., Szarvas, G., Vrbensky, R. (1996). *Tomato North-South: Recommendations for the improvement of environmental quality of tomato production using different environmental labels*. Centre of Environmental Science, Leiden University. Leiden. xxxi+96+31

Cowell, S.J. (1998). *Environmental Life Cycle Assessment of Agricultural Systems: Integration Into Decision-Making*. Ph. D. thesis. Centre for Environmental Strategy. University of Surrey. Guildford.

Jolliet, O. (1993). Bilan écologique de la production de tomates en serre. *Revue S. Vitic. Arboric. Hortic.* 25 (4): 261-267.

Jungbluth, N., Tietje, O., Scholz, R.W. (2000). Food Purchases: Impacts from the Consumers' Point of View Investigated with a Modular LCA. *International Journal of Life Cycle Assessment.* 5: 134-142.

Nienhuis, J.K., Vreede, P.J.A.d., De Vreede, P.J.A., Brumfield, R.G. (1996). Utility of the environmental life cycle assessment method in horticulture. En: *Proceedings of the XIIIth International Symposium on Horticultural Economics, Rutgers*. (Ed). New Brunswick, New Jersey, USA. No. 429: 531-538.

Van Woerden, S. (2001). The application of Life Cycle Analysis in glasshouse horticulture. En: *International Conference LCA in Foods*. (Ed). Gothenburg. 136-140.

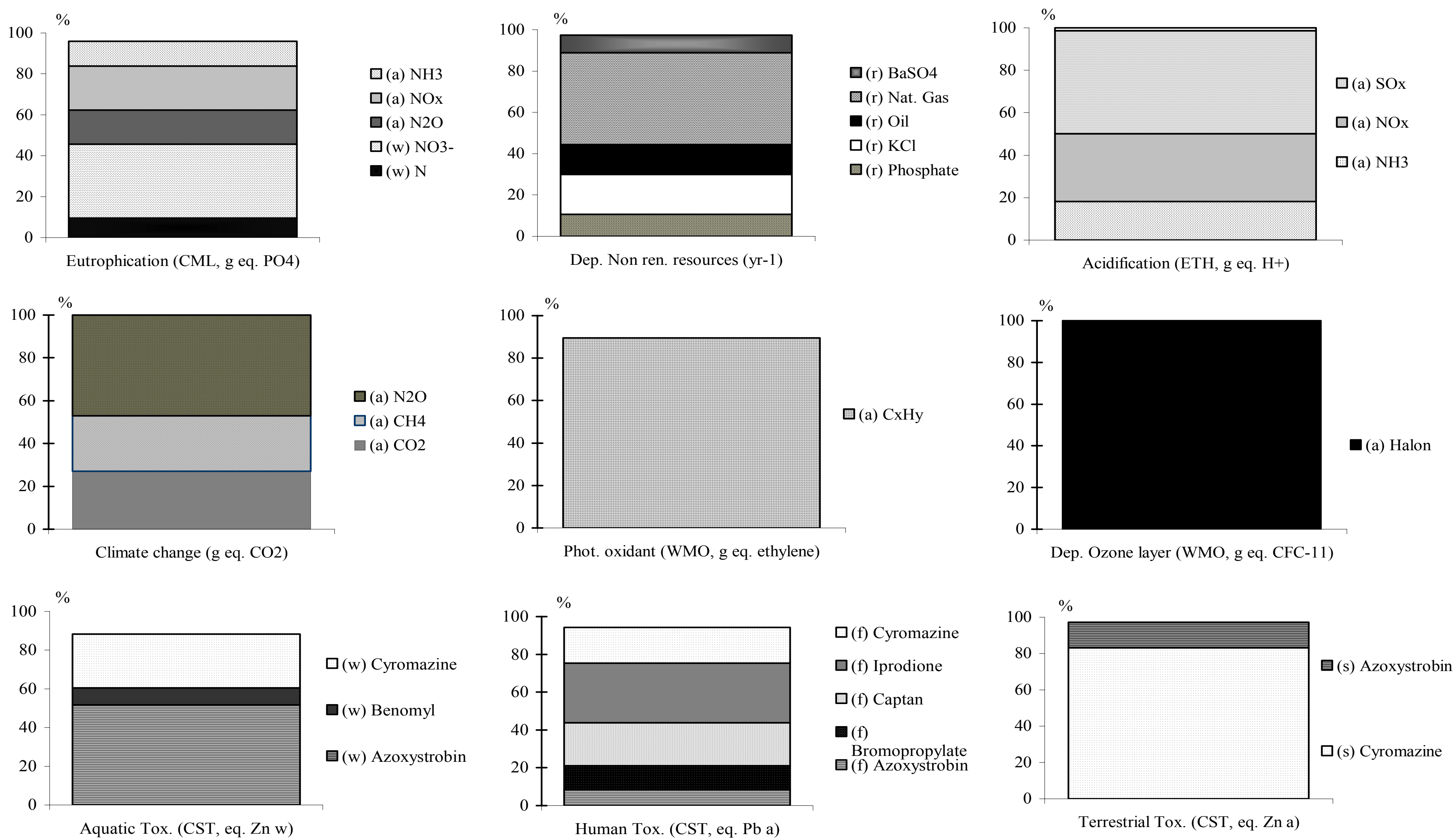


Figure 1. Main contributors' substances to the total impact categories

Table 1. Values for each one of the subsystems of the tomato crop production for the different environmental categories.

				FAB		PRODUCTION					WASTE
			TOTAL	G	R	GM	F	FR	P	PM	W
Eutrophication	EI	g eq. PO_4	2.7E-1	2.4E-2	7.3E-3	7.3E-3	4.2E-2	1.6E-1	9.0E-5	2.3E-4	2.6E-2
Dep. non renov. resources	BR	$year^{-1}$	8.1E-3	1.6E-3	6.1E-4	8.2E-4	5.3E-3	4.2E-5	1.1E-5	1.1E-5	-1.9E-4
Acidification	AI	g eq. H^+	3.0E-2	8.6E-3	3.3E-3	4.3E-3	7.1E-3	6.5E-3	7.3E-5	1.9E-4	3.5E-5
Climate change	CCI	g eq. CO_2	1.2E+2	4.0E+1	1.2E+1	-5.7E+1	9.7E+1	3.1E+1	4.2E-1	1.1E+0	-7.5E+0
Dep. ozone layer	ODI	g eq. CFC11	2.3E-5	2.3E-6	8.0E-7	1.6E-5	2.5E-6	4.7E-7	1.5E-7	1.2E-7	6.4E-7
Phot. oxidant	POI	g eq. ethylene	1.7E-1	7.7E-2	3.2E-2	4.1E-2	1.9E-2	1.4E-3	2.8E-4	3.7E-4	1.2E-3
Aquatic Tox	ATI	eq. Zn water	6.6E+0	2.0E-3	2.6E-4	1.7E-3	7.5E-4	9.2E-5	1.8E-5	6.3E+0	2.3E-1
Human Tox	HTI	eq. Pb air	4.9E+4	4.3E-1	1.2E-1	3.0E-1	3.4E-1	6.9E-2	6.5E-3	4.9E+4	1.3E-1
Terrestrial Tox	TTI	eq. Zn air	1.7E+2	4.6E-5	8.5E-6	2.3E-5	3.9E-5	5.0E-6	5.2E-7	1.7E+2	6.1E-4

Table 2. Main substances contributing to impact assessment.

Type	Substance	Category	Abbreviation
air	Ammonia	Acidification	AI
air	Carbon Dioxide	Climate change	CCI
air	Halon	Deplet ozone layer	DOI
air	Hydrocarbons (except methane)	Phot. oxidant	POI
air	Methane	Climate change, Phot. oxidant	CCI, POI
air	Nitrous Oxide	Climate change	CCI
air	Nitrogen Oxides	Eutrophication, Air Acidification	EI, AI
air	Sulphur Oxides	Air Acidification	AI
water	Nitrates	Eutrophication	EI
water	Nitrogen	Eutrophication	EI
resource	Barium Sulphate	Depletion non renovable resources	BR
resource	Oil (in ground)	Depletion non renovable resources	BR
resource	Natural Gas (in ground)	Depletion non renovable resources	BR
resource	Potassium Chloride	Depletion non renovable resources	BR
resource	Phosphate	Depletion non renovable resources	BR
Pesticides			
food, soil, water	Azoxystrobin	Human, Terrestrial and Aquatic Tox	HTI, TTI, ATI
water	Benomyl	Aquatic Tox	ATI
food	Bromopropylate	Human Tox	HTI
food	Captan	Human Tox	HTI
food, soil, water	Cyromazine	Human, Terrestrial and Aquatic Tox	HTI, TTI, ATI
food	Iprodione	Human Tox	HTI

Using LCA for the Improvement of Waste Management in Greenhouse Tomato Production

Muñoz, P.[(a)]; Antón, A.[(a)]; Montero, J.I. [(a)]; Castells, F.[(b)]

[a] *Institut de Recerca i Tecnologia Agroalimentària. Centre de Cabrils. Ctra de Cabrils s.n. 08348, Cabrils (Barcelona). Spain*

[b] *Department of Chemical Enginering. ESTEQ. Universitat Rovira Virgili 43007 Tarragona. Spain, pere.munoz@irta.es*

Abstract

Protected cultivation is aimed to obtain higher yields by modifying and improving natural climatic conditions. In spite of the fact that Mediterranean horticulture is mainly based on low-technology cold greenhouses, and therefore input resources are less than those used by more complex greenhouses, protected cultivation has a certain environmental impact. One of the bottlenecks associated to this system of production is the large amount of solid waste production. Waste can be biologic such as non-yield biomass and organic substrates, plastics (cover film, mulching etc…) or minerals (such as steel and mineral substrates). Life Cycle Assessment, LCA, has been used to assess different scenarios of waste treatment generated in a greenhouse tomato crop. The production of plastic waste from the different material used was estimated in 1750 kg ha^{-1} year^{-1}. Soilless closed systems, which reduce contamination from fertilisers and use of water, generated an estimated waste of 1150 kg ha^{-1} year^{-1} of polyethylene used in substrate bags and soil cover. Finally there is a non-yield biomass of 20000 kg of dry matter ha^{-1} year^{-1}. Different scenarios such as landfill, incineration and compost of the biomass have been evaluated. Results shows that for most indicators categories the waste management is important in the life cycle of greenhouse tomato production.

Keywords: compost, incineration and landfill

Introduction

European Union Landfill Directive (EC, 1999) requires their members to take measures to reduce biodegradable wastes going to landfills to 75% by 2004 and 35% by 2014. It is clear that the current recovery and recycling practices in Europe are not able to reach the targets set by this directive unless composting as a way of organic recycling would be performed at a commercial scale (Ren, 2003).

Meanwhile in recent years, this practice is being promoted for municipal solid waste and also for waste from livestock and gardens. For the other agricultural sectors composting activities are not well planned. Horticulture and fruticulture are important contributors to biomass waste, which is caused by non-harvested part of the plants when the crop is finished in horticulture, or by the pruning waste in fruticulture. For instance, for a tomato crop there could be a non-yield biomass next to 20000 kg of dry matter ha^{-1} year^{-1} (Stanghellini et al., 2003)

Also, greenhouse industry generates and important quantity of plastics waste. The production of plastic waste for renewing the polyethylene covers was estimated in 1000 kg ha^{-1} $year^{-1}$. Plastic waste from irrigation system is another 500 kg ha^{-1} $year^{-1}$, and in addition 250 kg ha^{-1} $year^{-1}$ generated from other materials such as film fastening bars, tutoring pincers, plastic strings, …(Antón, 2004). Closed system is a growing system where the water drained from the root zone is recollected and reused for irrigation of the same crop. Soilless closed systems, which reduce contamination from fertilisers and use of water, generate an estimated waste of 1150 kg ha^{-1} $year^{-1}$ of polyethylene used in substrate bags and soil cover.

This study uses LCA to compare different waste management for biodegradable matter and plastic waste from the horticultural sector. The aim is to promote and give diffusion to the importance of waste management in the global cycle of crop production to advance in the sustainable use of natural resources and pollution prevention following the waste policy of the European Union.

Material and Methods

Figure 1 shows the schematic process of the system under analysis, which was divided in three systems:

1. INF that includes production of greenhouse structure and components of the fertirrigation system
2. PROD production of the tomato crop, which take into account production of agrochemicals, fertilizers and pesticides, fertirrigation and integrated pest management
3. WASTE

Tomato was grown in a greenhouse with steel frame structure and LDPE cover film. Plants were grown in a closed system where the substrate was perlite bags.

Three scenarios of waste management were considered: **A)** biomass compost taking into account avoided environmental loads and plastic landfill., **B)** biomass and compost landfill and **C)** biomass and compost incineration. Data from landfill an incineration were obtained from DEAM database and corrected for non-yield biomass. Non-yield biomass production was evaluated as 0,044 $g_{\text{dry weight}}$ FU^{-1}. The composition of biomass was considered as $C_{27}H_{38}O_{16}N$ (Haug, 1993). It was accepted that the proportion of anaerobic decomposition of this biomass was the same as that of the decomposition of glucose (Soliva, personnel communication). Following this approach the anaerobic decomposition of one mol of non-yield biomass produced 594 g of CO_2 and 189 g of CH_4 (Antón, 2004). Data from compost plant were obtained from (AGA, 2002). To calculate the avoided environmental loads by using compost, data from Soliva (1998)and Rovira (1997) were used.

The following impacts categories, typically used in LCA, were assessed: CCI-Climate Change, (g eq. CO_2); WMO-Depletion of the ozone layer, (g eq. CFC-11), WMO-Photochemical oxidant formation, (g eq. ethylene); ETH-Air Acidification, (g eq. H^+), BR-

Depletion of abiotic resources, (yr^{-1}), CML-Eutrophication, (g eq. PO4). In greenhouse production the main contributions to toxicity categories are pesticides. In the waste subsystem toxicity categories are not so important in comparison with the total production system and therefore are not presented.

Results

Comparing scenario **B)** landfill to **A)** compost of the biomass, the subsystem WASTE in landfill disposal presents a 60 times higher score for the greenhouse effect, IPCC, and Photochemical oxidant formation, mainly caused by the methane emissions that comes from the decomposition of the biomass. This scenario is also 6.5 and 3.7 times higher for acidification and eutrophication respectively. Composting the biomass also represents a reduction on the depletion of non renewable resources due to the reductionon the use of chemical fertilizers (figure 2).

Comparation of scenario **C)** Incineration of waste, plastics and biomass to **A)** presented seven times higher impact score in greenhouse effect (climate change) and acidification. Depletion of the ozone layer and Photochemical oxidant formation showed ratios of 1.3. Besides, eutrophication had smaller effect in **C)** than in **A)**. Incineration produced human and ecosystems toxicities due to heavy metal emissions that there are not shown here because they are small compared to toxicity produced by pesticides (figure 2).

Impacts in Scenario **A)** are clearly smaller than **B)** or **C)** and they are caused mainly by the plastic landfill. Recycling plastic and the use of biodegradable plastics are subjects for future research.

Conclusions

- The main impact of the greenhouse tomato production is due to the waste of biomass and plastics. Especially categories such as climate change, eutrophication and photochemical oxidant formation are strongly influenced by the different treatments. Therefore suitable waste management is the best practicable environmental option.
- Compost of biodegradable matter is the best way of managing the waste to improve the impact assessment for most of the considered impacts categories.

Acknowledgements The authors thank Julio Rodrigo, Rafi Caceres and Montse Soliva for providing useful information related to compost process. This research was partially supported by INIA nº SC00-080-C2 and nº RTA03-096-C5-2.

References

AGA. 2002. Avaluació ambiental de diferents estratègies per a la gestió dels residus municipals ordinaris (RMO). Grup AGA-Centre d'Innovació SIMPPLE-STQ-URV i Junta de Residus (Departament de Medi Ambient).

Antón, A. 2004. Utilización del Análisis del Ciclo de Vida en la Evaluación del Impacto Ambiental del cultivo bajo Invernadero Mediterráneo. Tesis doctoral. Programa d'Enginyeria Ambiental. Universitat Politècnica de Catalunya. Barcelona.
EC. 1999. Landfill Directive. European Parliament and Council Directive 1999/31/EC of 26 April.
Haug, R. 1993. The practical handbook of compost engineering. . Boca Raton, Florida. Lewis Publishers.
Ren, X. 2003. Biodegradable plastics: a solution or a challege? Journal of Cleaner Production. 11: 27-40.
Rovira, S. 1997. Composició de l'extracte aquos d'un sòl adobat amb residus orgànics. TFC. Escola Superior d'Agricultura de Barcelona.
Soliva, M. 1998. Residus orgànics per a l'agricultura: un tema de recerca a l'escola superior d'agricultura de Barcelona (ESAB). Arxius de l'Escola Superior d'Agricultura, Sèrie cinquena, Nº 1. Barcelona.
Stanghellini, C., Kempkes, F.L.K., Knies, P. 2003. Enhancing Environmental Quality in Agricultural Systems. Proc. Proceedings of the International Symposium on Managing Greenhouse Crops in Saline Environment. International Society Horticultural Science. Acta Horticulturae. Pisa. Italy. 609: 277-283.

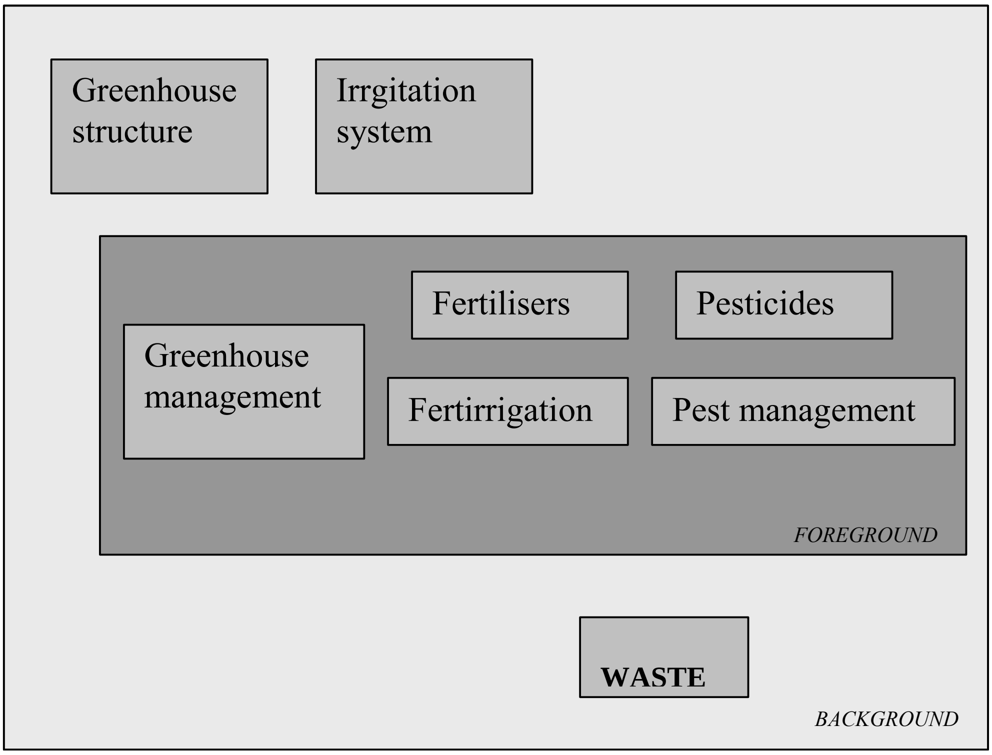


Figure 1. Schematic processes diagram of the system under study.

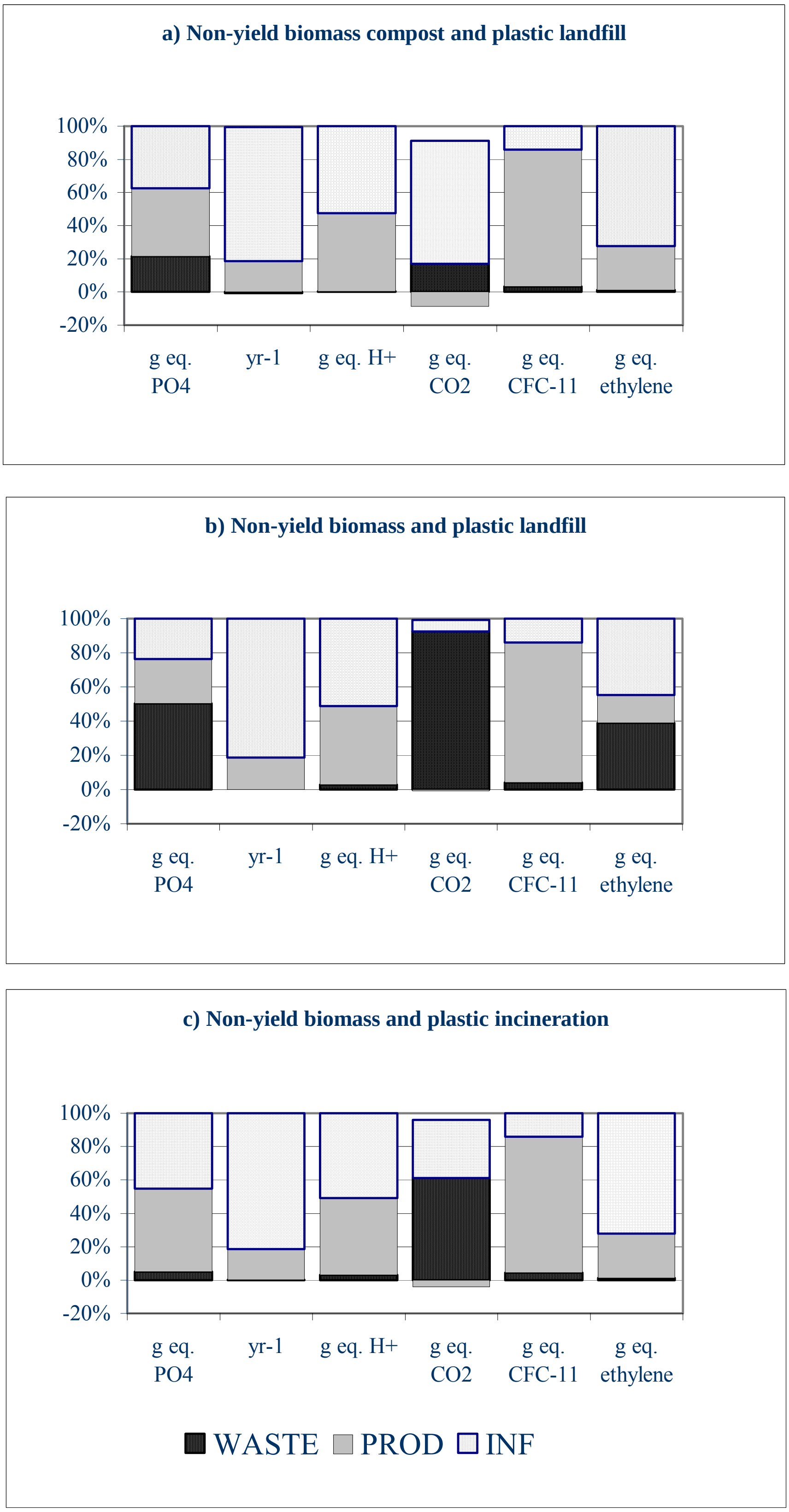


Figure 2. Influence of different waste management, WASTE, in six impact categories related to subsystems PROD, crop cultivation, and INF, production of estructure.

LCA of the integrated production of oranges in the Comunidad Valenciana (Spain)

N. Sanjuán*, G. Clemente and L. Úbeda
*Department of Food Technology. Universidad Politécnica de Valencia. Camino de Vera s/n. 46021Valencia. Spain. Tel. (+34) 963879366. Fax (+34) 963879839. nsanjuan@tal.upv.es

Abstract
An LCA of integrated orange production in the Comunidad Valenciana *was performed. The functional unit was 1kg of oranges. The production of agrochemicals, the production and use of energy in agriculture (for watering and machinery), and agricultural practices were studied. Hot spots were detected. The lack of environmental information for agricultural LCA in Spain and the need to adapt certain methodological aspects to Spanish soil and climate characteristics have also been pointed out.*

Key words: oranges, integrated farming, LCA

Introduction
Spain is the fourth largest orange producer in the world, with 5.4 million t of oranges harvested in 2000, of which ca. 3.6 million t are produced in the *Comunidad Valenciana*. Around 70% of the citric fruits of this region are exported, mainly to the European Union. Since the 1980s, both the state and regional governments have encouraged integrated farming to develop more sustainable agricultural practices. In the year 2000, more than 60% of the registered orange cultivated areas and around 65% of the total production were a result of integrated production (IP).

The main characteristic of IP is the strict control of agricultural practices that is carried out in order to optimize available resources and technologies. Nevertheless, it is important to have a holistic view of the agricultural processes to avoid the displacement of environmental problems from one part of the cycle to another. In this sense, LCA has proved to be a useful tool.

Goals and scope definition
The objective of this study is to carry out an LCA of the integrated production of oranges in the *Comunidad Valenciana* and the purpose is twofold: to evaluate the environmental impact of the agricultural practices carried out in IP of oranges in the *Comunidad Valenciana*, and to contribute to developing the application of LCA methodology to agricultural practices in Spain. The functional unit was 1kg of oranges leaving the farm gate.

Description of the system under study. System boundaries
In this study the agricultural production of oranges (variety *navelina)* according to IP system (DOGV, 2001) in the *Comunidad Valenciana* corresponding to the year 2000 was consid-

ered, taking into account a farm in full production (adult trees) with 500 trees/ha, a plantation frame of 4x5 m and an average yield of 30,000 kg oranges/ha.

From among the different systems of watering, drip irrigation using well water was studied. An average annual water volume of 5000 m^3/ha and 85% irrigation efficiency was taken into account. No tillage was done. Four herbicide treatments were carried out each year: the first in February with a commercial mix of glyphosate (18% w/v active ingredient) and MCPA (18% w/v a.i.); the second in March with glyphosate (36% w/v a.i.); the third in May, again using a commercial mix of glyphosate (18% w/v a.i.) and MCPA (18% w/v a.i.); and the last in August with glyphosate (36% w/v a.i.). Trees were pruned by hand and according to the IP normative, the ground pruning was left on the soil surface. The following solid soluble fertilizers were applied together with water: ammonium nitrate (33.5% de N), phosphoric acid (54% P_2O_5) and potassium sulphate (13% N y 46% K_2O). The applied rates were 782, 120 and 293 kg/ha respectively. Once a year sheep manure was applied (3600 kg/ha).

The stages considered were agrochemical production (fertilizers and pesticides), production of energy directly used in agriculture (for machinery and watering system) and agricultural practices. The inventory was based on data for average farm practices provided by FECOAV (*Federación de Cooperativas Agrícolas Valencianas*). Data regarding ammonium nitrate and phosphoric acid production have been obtained from DEAM database, and data for potassium nitrate were based on Davis and Haglund (1999). Emissions that originate in manure production were not included because they were allocated to the manure producer. The emissions from the input of fertilisers to the agricultural soil were obtained from a nutrient balance. Data on energy consumption for pesticide production were from Green (1987) and, when the active ingredient was not available, the extrapolation method proposed by Audsley *et al*. (1997) was carried out. Transport of agrochemicals to the farm was not included. Although these products are mostly formulated in Spain, their active ingredients are produced in many different countries.

The production of capital goods (agricultural machinery, watering pumps and buildings) was not included as they have a long life. Data regarding energy consumption of agricultural machinery were obtained from a study carried out by Gracia *et al*. (1986) into orange cultivation practices in Valencia. Although these data are not current, it can be considered that there has been little change in the practices over the years given the large number of existing small farms. The energy for watering was computed from the pressure and the volume of water needed. Data on energy production was obtained from the aforementioned DEAM database.

Impact categories
The impact categories selected were: acidification, non-renewable resources depletion and eutrophication, according to the CML method; global warming and photochemical oxidant formation using the WMO method; ozone depletion following the POPC method; and terrestrial and human toxicity, with the USES method.

Results
From the impact assessment results (Figure 1) it is clear that the fertilizer production greatly contributes to acidification (86% of total impact), mainly due to ammonia emissions in ammonium nitrate production, and to non-renewable resources depletion (84%), attributable to rock phosphate, potassium chloride and natural gas consumption. Fertilizer production also contributes to the greenhouse effect (52%) and photochemical oxidant formation (42%) given the emissions from combustion processes, and ozone depletion (25%) mainly due to Halon 301 formation during the combustion processes in the presence of F and Br. The production and use of energy for watering and agricultural machinery also plays a decisive role in photochemical oxidant formation and ozone depletion (33% and 41% of total impact, respectively). The agricultural practices are what mainly lead to eutrophication (99.9%), due to nitrate leaching. These practices mostly responsible for human and terrestrial toxicity (96 and 85%, respectively) mainly due to the use of copper as fungicide and also to 31% of the greenhouse effect. The production of pesticides causes 15% of the depletion of non-renewable resources (mainly because of the extraction of Cu), 34 % of ozone layer depletion and 25% of photochemical oxidant formation, the latter two impacts being the result of fuel combustion.

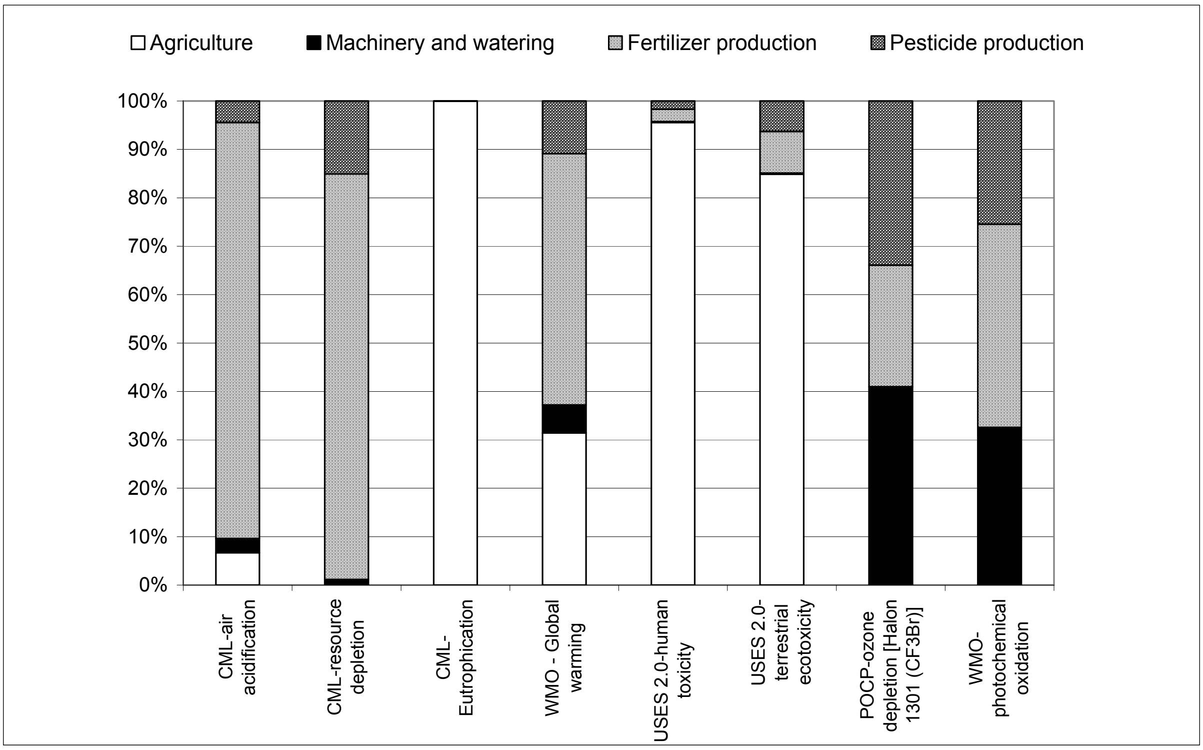


Figure 1. The environmental profile of IP oranges in Valencia.

Discussion

Hot spots and improvements

Nowadays, the eutrophication caused by agricultural practices is one of the main problems in Valencia. In order to prevent this, nutrient balances should be performed. Regarding the impact of fertilizer production, a comparative study of the manufacturing process of fertilizers is needed to determine if there are significant differences in the efficiency of use of resources and the emissions caused in the production process. In order to reduce the depletion of rock phosphate alternative sources of P should be applied, taking also into account that in Valencian soils, P is mostly immobilized.

Methodology problems

One of the critical problems when performing this LCA was the lack of data, mainly with respect to fertilizers, pesticides and machinery production. Other important aspects are the emissions derived from manure. This lack of data has obliged us to use those agrochemicals available in the databases instead of selecting, in specific cases, the most representative ones.

As for the methodology, it is also important to point out that the evaluation methods for toxicity consider only specific soil and climate characteristics. For example, in this LCA, copper contributed greatly to toxicity impact categories, yet we must bear in mind that given the characteristics of Valencian soils (basic pH), copper is retained and its toxic effect decreases.

References

Audsley, E. (coordinator) Alber, S., Clift, R., Cowell, S., Gaillard, G., Hausheer, J., Jolliett, O., Kleijn, R., Mortensen, B., Pearce, D., Roger, E., Teulon, H., Weidema, B. and van Zeirs H. 1997. Harmonisation of Environmental Life Cycle Assessment for Agriculture. Final Report. Concerted Action AIR3-CT94-2028. Silsoe Research Institute. Bedford, United Kingdom.

Davis, J. and Haglund, C. 1999. Life Cycle Inventory (LCI) of Fertiliser Production. Fertiliser Products Used in Sweden and Western Europe. SIK report nº654. Chalmers University of Technology. The Swedish Institute for Food and Biotechnology (SIK). Gothenburg, Sweden.

DOGV (Diari Oficial de la Comunitat Valenciana). 2001. 3 de gener de 2001. RESOLUCIÒ de 23 de novembre de 2000, del director general d'Innovació Agrària i Ramaderia, per la qual s'estableixen les normes per a la Producció Integrada en cítrics en l'àmbit de la Comunitat Valenciana. [2000/9741].

Gracia, C., Val, L. and Juste, F. 1986. Consumo Energético de las Operaciones Mecanizadas en el Cultivo de los Agrios. *Investigación Agraria. Producción y Protección de Vegetales*. **1** (2), 185-207.

Green, M. 1987. Energy in Pesticide Manufacture. Distribution and Use. In: Helsel, Z. (ed.) *Energy in Plant Nutrition and Pest Control*. Elsevier. Amsterdam, The Netherlands.

Environmental Impact Assessment at Commercial Dairy Farms

Comparison of LCA, ecological footprint analysis, and input-output accounting

De Boer, I.J.M., Iepema, G., and Thomassen, M.A.*

Animal Production Systems Group, Wageningen Institute of Animal Sciences
PO Box 338, 6700 AH Wageningen, Wageningen University, Wageningen, The Netherlands

Abstract

The aim of this article was to determine effectiveness of environmental indicators obtained from input-output accounting (IO), Ecological Footprint Analysis (EFP) and Life Cycle Assessment (LCA) to show differences among commercial dairy farms. In total, 11 environmental indicators were quantified and correlated. Results show that input-output accounting of nutrients yields effective indicators with respect to eutrophication and acidification. Regarding land and energy use, however, EFP and LCA yield similar indicators. EFP, subsequently, sums land and energy use into one final unit, i.e., biologically productive area, which is shown to be ineffective. LCA appears a useful technique to deduce effective, environmental indicators. LCA results with respect to acidification potential, for example, showed that on-farm NH_3 emission per ha or per kg milk is an effective indicator to show differences among production systems and to improve performance with a system.

Key words: organic dairy production, environmental impact assessment, input-output accounting, ecological footprint analysis, life cycle assessment.

Introduction

To show differences in environmental impact among production systems, such as for example organic and conventional milk production, environmental impact should be assessed at a large number of commercial farms for each production system of interest. In principle, two different approaches can be used to assess the environmental impact at commercial agricultural farms. First, the input-output accounting approach (IO), which computes the difference in nutrients entering and leaving the farm gate, while the farm itself is considered a black box. Second, a “cradle to grave or farm-gate” approach, which computes the integrated environmental impact of an agricultural activity throughout its entire life cycle, such as Ecological Footprint analysis (EFP) and Life Cycle Assessment (LCA). The aim of this article is to determine the effectiveness of environmental indicators deduced from IO, EFP and LCA, to show differences among commercial dairy farms.

Material and methods

Data. The environmental impact of eight commercial organic dairy farms was assessed, using IO, EFP, and LCA. These eight farms participated in a demonstration project of Dutch organic dairy farmers, the so-called BIOVEEM project (Smolders and Wagenaar, 2004). In this project, a large number of on-farm data are gathered regularly in order to improve farm performance. Additional on-farm and off-farm data were required, which were obtained by questionnaires and expert consultation.

Input-output accounting. For each dairy farm, we computed the annual surplus of nitrogen (N) and phosphate (P_2O_5) using input-output accounting at farm level. This surplus is derived from the difference between farm inputs and farm outputs, with respect to N and P_2O_5, while the farm itself is considered a black box (Ondersteijn et al., 2002). Farm inputs of N and P_2O_5 considered were: clover N-fixation, deposition, and import through roughage, concentrates, animals and manure. Farm outputs of N and P_2O_5 considered were: animal products, i.e., milk, meat, manure or living animals, and plant products, i.e., roughage or crops. The difference between inputs and outputs is called the farm surplus, and is assumed to be lost to the environment. This farm surplus is expressed per ha farm area. To further diversify the N surplus, we first corrected this surplus for annual NH_3 emission. The corrected N surplus per farm area, therefore, includes environmental losses of NO_x, N_2O and NO_3^- only. This approach, therefore, yields three indicators, i.e., NH3 emission (kg N) per ha, surplus of N (kg) per ha, and surplus of P_2O_5 (kg) per ha.

Ecological Footprint Analysis. For each dairy farm, an ecological footprint (EFP) was computed. A farms EFP is the biologically productive area (BPA) needed to produce all resources and to absorb waste (i.e., CO2 from fossil fuel combustion) generated by that farm (Wackernagel and Rees, 1996). To compute EFP of a dairy farm we, therefore, keep track of land and energy requirements of all resources imported into the farm or used on the farm. Subsequently, land use and energy use is summed into BPA, assuming that 1 ha of wood land absorbs all CO_2 released during combustion of 100 GJ of energy. To determine BPA of co-products, such as feed ingredients, economic allocation is used. Finally, a farms BPA is allocated to milk production, based on economic allocation, and expressed per kg of Fat and Protein Corrected Milk (i.e., kg FPCM = (0.337 + 0.116×%fat + 0.06×%protein)×kg milk production (CVB, 2000). EFP, therefore, yields one indicator, i.e., BPA (m^2) per kg of FPCM.

Life Cycle Assessment. For each dairy farm, a “cradle to farm-gate” LCA of one kg of FPCM was performed (de Boer, 2003). The following impact categories were assessed: land use, energy use, global warming potential (GWP; CO_2, CH_4, N_2O), eutrophication potential (EP; NO_x, PO_4^{3-}, NO_3^-, NH_3, NH_4^+) and acidification potential (AP; SO_2, NO_x, NH_3). Economic allocation was used. LCA, therefore, yields the following environmental indicators: ha land use/kg FPCM, MJ energy use/kg FPCM, GWP in CO_2-eq/kg FPCM, EP in NO_3^- eq/ha or in NO_3^-eq/kg FPCM, AP in SO_2 eq/ha or in SO_2 eq/kg FPCM.

Results and discussion

Input-output accounting. For each farm, results of environmental indicators computed are given in Table 1. The average N surplus was 82.5 kg per ha, with a standard deviation of 61.6. Variation in N surplus mainly was due to variation in clover N-fixation and import of roughage. The average P_2O_5 surplus was 5.4 kg per ha, with a standard deviation of 16. This extreme variation was due to the fact that halve of the farms had a negative and half of the farms a positive P_2O_5 surplus.

Ecological footprint analysis. BPA of an organic dairy farm on average was 1.85 m^2 per kg of FPCM, with a coefficient of variation of 18%. EFP sums land use and energy use in one final unit (i.e., BPA) by assuming that 1 ha of woodland fixes all CO_2 released during combustion of 100 GJ of energy. For farm no. 1, for example, BPA/kg FPCM is 1.39, which is the sum of 1.21 ha due to actual land use and 0.18 ha required for CO_2 absorption due to energy combustion (see Table 1). No correlation, however, was found between BPA from land use and BPA from energy combustion for eight farms studied. Summation of BPA from land use and BPA from energy combustion, therefore, implies loss of information regarding the environmental performance of commercial farms, and, therefore, seems inappropriate.

Life cycle assessment. An organic dairy farm used an average of 1.6 m^2 land per kg FPCM. From this, 69% is on-farm land, such as grassland and arable land, whereas 31% is off-farm land required mainly for cultivation and transport of feed. An organic dairy farm used an average of 2.48 MJ energy per kg FPCM. From this energy use, 40% is direct, on-farm energy consumption, whereas 60% is required for the production of farm inputs like concentrates, purchased roughage and extern labour. GWP of an average farm was 1.81 kg CO_2-eq/kg FPCM, of which 78% is due to on-farm emission of CH_4 and N_2O mainly, each explaining around 50%. EP of an average farm was 82.1 g NO_3^- eq/kg FPCM or 720.3 kg NO_3^- eq/ha, of which around 50% is due to on-farm emission of mainly NO_3^-, $PO_4^-$and NH_3. Off-farm EP is explained mainly by cultivation and transport of feed. AP of an average farm was 104 kg SO_2 eq/ha or 11.8 g SO_2 eq/kg FPCM. AP is for around 70% due to on-farm emission of mainly NH_3.

Correlations between environmental indicators. As described previously, EP per ha was for 47% due to on-farm leaching of NO_3^-, PO_4^- and emission of NH_3. A correlation was found between the LCA indicator EP per ha (i.e., g NO_3^- eq per ha) and the input-output accounting indicator N surplus per ha (correlation 0.8; p=0.01) and between EP per ha and P_2O_5 surplus per ha (correlation 0.75; p=0.03). Similarly, a correlation was found between AP and NH_3 emission. Expressed per ha farm area, AP and NH_3 show a correlation of 0.92 (p=0.001). AP per ha farm area also correlated to Dutch Livestock Units (LU) per ha (0.92; p=0.001), which is due to the fact that computation of NH_3 emission used largely depends on animal numbers. No correlation, however, was found between fossil fuel use and GWP per kg FPCM (correlation 0.2; p=0.6), which is due to the fact that CH_4 and N_2O and not CO_2 are main contributors to GWP.

Conclusion

Unlike input-output accounting, ecological footprint analysis and especially LCA of milk production at commercial dairy farms appears time-consuming. LCA results, however, show insight into the environmental impact of various processes in the chain of milk production. This insight is highly relevant to deduce environmental indicators necessary to show differences in environmental impact among production systems. Regarding land and energy use, e.g., ecological footprint analysis and LCA yield similar indicators. Regarding AP, for example, a comparison of dairy production systems could be based on the indicator on-farm NH_3 emission per kg FPCM or per ha farm area. Regarding EP for current dairy production systems, N and P surplus per ha are indicators that detect variation among and within production systems.

References

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

CVB, 2000. Table book animal nutrition; Livestock norms and feeding values. Central Buri for Animal, Lelystad, The Netherlands.

Ondersteijn, C.J.M, A.C.G. Beldman, C.H.G. Daatselaar, G.W.J. Giesen, Huirne, R.B.M. 2002. The Dutch Mineral Accounting Systems and the European Nitrate Directive: Implications for N and P management and farm performance. Agriculture, Ecosystems and Environment 92:283-296.

Wackernagel, M. W. Rees. 1996. Our ecological Footprint. Reducing Human Impact on the Earth. New Society Publishers, Canada.

Smolders, G., Wagenaar, J.P., 2004. BIOVEEM. Practice and research on 10 organic dairy farms 1997-2001. Applied Research, Animal Sciences Group, Wageningen University and Research Center, Lelystad (in press).

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Table 1. Results of environmental indicators obtained from input-output accounting, ecological footprint analysis and life cycle assessment for eight organic dairy farms studied.

	Farm 1 to 8								
Environmental indicators	1	2	3	4	5	6	7	8	Mean (SD)
Input-output accounting									
kg N surplus/ha	47.3	197.3	49.1	87.3	152.8	25.0	31.8	69.7	82.5 (61.6)
kg NH_3/ha	35.1	43.8	20.4	32.8	30.4	40.6	25.4	42.2	33.8 (8.3)
kg P_2O_5 surplus/ha	-4.3	21.5	12.3	-0.5	29.7	-15.3	-12.0	11.8	5.4 (16.0)
Ecological Footprint									
m^2 BPA/kg FPCM	1.39	2.38	2.02	1.79	2.07	1.37	1.85	1.91	1.85 (0.33)
• land	1.21	2.11	1.71	1.53	1.64	1.19	1.72	1.68	1.60 (0.30)
• energy combustion	0.18	0.27	0.31	0.26	0.43	0.19	0.13	0.23	0.25 (0.09)
Life Cycle Assessment									
land use (m^2/kg FPCM)	1.21	2.11	1.71	1.53	1.64	1.19	1.72	1.68	1.60 (0.30)
energy use (MJ/kg FPCM)	1.78	2.65	3.11	2.55	4.26	1.88	1.31	2.31	2.48 (0.91)
GWP (kg CO_2 eq/kg FPCM)	1.17	1.77	1.64	3.85	1.76	1.16	1.56	1.60	1.81 (0.86)
EP (g NO_3^- eq/kg FPCM)	65.0	101.4	74.4	97.6	153.3	34.3	37.7	93.4	82.1 (38.6)
EP (kg NO_3^- eq/ha)	756.1	1068.5	449.1	977.7	1132.8	357.8	187.2	833.2	720.3 (350.7)
AP (g SO_2 eq/kg FPCM)	10.1	15.0	9.8	11.0	12.4	10.3	10.9	15.0	11.8 (2.14)
AP (kg SO_2 eq/ha)	117.0	158.4	59.0	110.2	91.7	107.8	53.9	134.1	104.0 (35.4)

GWP = Global Warming Potential, EP = Eutrophication Potential, AP = Acidification Potential.

A systematic description and analysis of GHG emissions resulting from Ireland's milk production using LCA methodology

*Casey, J.W**, *Holden, N.M*

Department of Agricultural and Food Engineering, University College Dublin, Earlsfort Terrace, Dublin 2, Ireland. Tel. +353(1)7165557 Fax: +353(1)4752119, john.casey@ucd.ie

Abstract

Life Cycle Assessment (LCA) methodology was used to assess the greenhouse gas (GHG) emissions resulting from rotational grazing for dairy production in Ireland. A system definition was formulated with average herd size (47 cows), total inorganic nitrogen usage (175 kg N ha^{-1}), and concentrate consumption (818 kg animal yr^{-1}) estimated from national data. Mechanisation inputs were also included. System processes that contribute to GHG emissions are identified. Emission factors were ascribed to the identified GHG contributors to assess the effect of 1litre of milk production (FU) on climate change.

Keywords: *GHG emissions, LCA methodology, dairy production, Ireland*

Introduction

Greenhouse gas emissions attributable to dairy farms arise primarily from three main sources - methane (CH_4) from livestock, nitrous oxide (N_2O) from pasture improvement and carbon dioxide (CO_2) from energy generated by the combustion of fossil fuels. The objectives of this study were to: (i) produce a system inventory for all input and outputs; (ii) identify GHG contributers to the system; and (iii) assess GHG emissions per functional unit (FU) using part of the LCA methodology.

System description

The systems boundary is defined by the environmental burden of Irish milk production with respect to GHG emissions. Political boundaries are not considered as limits to the system. Distribution and milk processing are beyond the scope of the boundary. In Ireland over 1.2 m dairy cows on *c.* 27,000 farms consume grass in the field or as conserved forage, with additional feed as concentrates (40% home grown and 60% imported). Weather permits 220-240 days grazing and housed feed is either grass silage and/or forage maize apart from concentrates. The system can operate over a range of farm sizes to produce 5.55 x 10^9 L of milk quota (European Community, 2003).

Materials & Methods

Data were collected from National Farm Surveys and the Fertiliser Survey (all produced by Teagasc the Irish Agricultural and Food Development authority) and by personal communications. It was assumed that all manure produced on the farms is used as fertiliser and nitrogen excretion data were taken from Smith and Frost (2000). A system inventory was constructed (Table 1) and for each element a GHG emission (scaled to CO_2 as a reference) is being allocated (IPCC, 1996). The defined FU is: the production of national milk quota in litres, scaled to the output of one litre over a time frame of one year (total greenhouse gas emissions per functional unit – TGE/FU). The global warming potential (GWP) index will be used to assess the system for total GHG emissions per FU.

Results

The Irish system can be summarized as follows: average herd size is 47 cows, mean fertilizer applied is 175 kg N ha^{-1}, the mean concentrate used is 819 kg cow yr^{-1} and the average milk output is 4822 L cow yr^{-1}. The sources of GHG emissions are outlined in Table 1. The calculated TGE/FU is 1.36.

Discussion and Conclusions

To assess Irish milk production for GHG emissions it was thought that the LCA methodology could be partly used. However there are a number of limitations associated with applying the LCA methodology to agricultural production. For example using the LCA framework for the study suggested that elimination of processes with small emissions would make a more robust and reliable inventory. However the project set out to assess the entire environmental burden of milk production in terms of GHG emissions irrespective of the magnitude of the emission. Another issue is that most of the emission data are not from Ireland. The data are mainly UK, German, Danish and Dutch. For example dung emissions were taken from UK data and manipulated to suit the Irish system. A figure of 1.36 TGE/FU is presented from preliminary examination of the system not including the contributions of concentrate feed emissions and diesel usage associated with silage making.

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References

European Community, 2003. COMMISSION REGULATION (EC) No 572/2003.

IPCC, Intergovernmental Panel on Climate Change, 1996. Climate Change 1995. The Science of Climate Change. Contribution of Working Group I to the Second Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, UK.

Smith, K.A. & Frost, J.P., 2000. Nitrogen excretion by farm livestock with respect to land spreading requirements and controlling nitrogen losses to ground and surface waters. Part 1: Cattle and sheep. *Bioresource Technology* **71**, 173-181.

Table 1. Identified GHG contributors (Preliminary), kgs CO_2 equivalents.

Category	Input	Output	Methane	Carbon dioxide	Nitrous Oxide
Size in ha	40				
Cows in milk	47.2		4720		
Other stock	69.7		3485		
Stocking rate (LU ha^{-1})	1.8				
Milk Output (L cow yr^{-1})		4822			
Total litres per farm		227598			
Farm management					
Fertiliser production (kg N)	7000		12	18620	94
Importation (fert. km)	1040		0.36	234	0.05
Merchant to farm (fert. km)	40		0.32	208	0.05
Fertilizer applied (kg N ha^{-1})	175				85
Diesel used (kg)	3204		2	11406	2
Electricity (kwh)	8338			6504	
Manure management					
Slurry indoors Storage			257		4
Dung in field			0.001		48
Collecting yard			0.002		0.00004
Spreading			7.7		20
Other cattle					
Slurry indoors Storage			194		2.5
Dung in field			0.04		20
Spreading			5		13
Total kg CO2 eqv.			182362	36971	89684
Total			309017		
TGE/FU			**1.36**		

Life cycle assessment results and related improvement potentials for oat and potato products as well as for cheese

Katajajuuri, J.-M.[a,*], Virtanen, Y.[a], Voutilainen, P.[a] and Tuhkanen, H.-R.[b]
[a]MTT Agrifood Research Finland, Environmental Management, FIN-31600 Jokioinen, FINLAND
[*]juha-matti.katajajuuri@mtt.fi, Phone: +358-(0)3-41881, Fax: +358-(0)3-4188 2222
[b]MTT Agrifood Research Finland, Karilantie 2A, 50600 Mikkeli, Finland

Abstract

LCA studies were performed for oat flakes, potato flour, hard cheese and IQF (Individually Quick-Freeze) potato gratin with cheese sauce produced in Finland. The main objective of the study was to compile reliable environmental impact data for all stages of the current production and supply systems and to identify and evaluate improvement potentials of them. No comparisons between products were made. Data for the system models were acquired from the field thus providing a reliable basis to analyse the sources of environmental impacts ("hot spots") and to consider respective improvement possibilities. In the cheese production system, special attention was paid on the management of complicated nutrient flows and wash-outs at the farms. As one part of improvement assessment, the effect of animal husbandry practices, like different feeding strategies, on environmental impacts in the entire cheese system were studied.

Keywords: Life Cycle Assessment, food, agriculture, field oriented data acquisition, improvement assessment.

Background

Finnish agricultural and food industry and trade, in co-operation with Finnish research institutes MTT and VTT, started in early 2000 a process to produce reliable environmental performance data on relevant Finnish food production and supply systems. This joint national effort was expected to support R&D and innovation activities in food production and had a further aim to improve the environmental performance of products according to the novel societal responsibility principle and integrated product policy (IPP).

So far the studies have covered feed barley and oat, silage, pasture, dry hay, feed concentrates, potatoes, oat, milk and cheese. The studies were a part of Environmental Cluster Research Programme and national quality strategy for food in Finland, which covers the entire food chain from farm inputs to the consumer. The studies were financed by the Ministry of Environment and Ministry of Agriculture and Forestry, as well as by the participating companies and the research organisations MTT and VTT. The main objectives of the studies have been to compile reliable environmental impact data for all process stages involved in the systems considered, to identify and assess the significance of the impact sources ("hot spots"), and to draw and evaluate possibilities to improve the environmental performance of the systems. The central data on the current supply webs were based on empirical investigation of the real processes. One important aim behind this was to get the different parties involved in the supply webs to learn more about product oriented environmental management, respective assessment of environmental impacts, and related benefits, i.e. learning by doing. This gives a real possibility to seek continuous improvements in the supply chains. Principles and benefits of supply web management based LCA are more widely presented and discussed by Virtanen and Poikkimäki (2003) and Poikkimäki and Virtanen (2003).

Methods and field oriented data acquisition

The central data was based on empirical investigation of the real processes. Mainly the parties of the supply chains did fieldwork for the data acquisition. Hence, data from industrial processes came from Finnish plants producing and processing raw materials and products needed in the systems. Average Finnish grid electricity data was used for electricity. Local energy production, like steam and utilisation of biogas from landfills etc, was taken as it currently appeared in the systems. Consumer products were assumed to be delivered over Finland according to the current regional market shares. Emissions of delivery webs were modelled using realistic delivery routes with initial loading, retail stops, and final discharge of return load each. Logistics was modelled in collaboration with Finnish logistics companies, including retail product losses. The data for retail refrigeration and freezing was drawn using nominal electricity consumptions of the refrigeration devices and assessed average product throughputs of the cold stores. Household cooking (oat meal and IQF gratin) and freezing (IQF gratin) were assessed based on appliance consumptions and respective times.

Milk production data and detailed feeding values were drawn from about 700 farms. Data acquisition for farm cultivation was based on individual farm level cropping plans and realization of these, and on interviews of farmers. Washouts (N and P) and air emissions (NH_3, CH_4, N_2O) were calculated on the basis of nutrient balances, nationally applied P washout models, IPCC reference manual (1997) for greenhouse gases and Finnish agricultural knowledge at MTT. Data acquisition methods and sources are described in detail in project reports (Voutilainen et al. 2003a, Katajajuuri et al. 2003, Voutilainen et al. 2003b.) Tools to handle input-output data collection electronically and generation of LCI data for farm level are under construction in Finland. Technique to be used of that might be a part of Finnish quality databank (see e.g. Katajajuuri & Loikkanen, 2000 or http://www.mmm.fi/english/agriculture/food_quality.htm). National IO tables were considered too generic for the assessments, and for seeking real improvements in particular chains and farms. The principal starting point was to model systems as they currently were. Because the systems studied were quite complicated with several by-products and various secondary outputs, allocation couldn't be completely avoided. Allocation principles utilised have been discussed in Katajajuuri and Voutilainen (2002). However, it was found that allocation principles have a big effect on the results, and in the future avoiding allocation through system expansion e.g. for milk and meat production should be carried out as proposed by Cederberg and Stadig (2001) and Weidema (2001). This will be especially important when making comparisons with different kinds of foodstuffs. The management of the data uncertainties and the ranges of data variation is an important research area, especially in agricultural systems. For this reason the development of quantitative uncertainty assessment was also included in the study using the gratin case for testing. Stochastic modelling, performed by Monte Carlo and Latin Hypercube simulation methods, was chosen as techniques for uncertainty assessment (see Voutilainen et al. 2003b).

LCA results in general terms

Nitrogen and phosphorus emissions to water and the respective eutrophication potentials of the studied systems were nearly all due to the nitrogen and phosphorous emissions from cultivation, and in milk production also due to washing of milk storage tanks and milking equipment on dairy farms. Other processes of the systems had very little influence on them. Instead, energy-related emissions, like carbon dioxides, varied considerably depending on the characteristics of many processes of systems, such as the fuels assumed for energy production, degrees of industrial refining, yields of processing, and need and efficiencies of freezing.

The contributions of the different production stages to acidification and global warming potentials (GWP) differ substantially between the product systems. In cheese system, when dairy cattle involved, acidification and GWP were dominated by the milk farm: crop and milk production. The case studies clearly showed that the environmental impacts of the systems are e.g. case-, allocation principle-, and production system dependent. Therefore, generalisation of the contributions of the life cycle stages to other food chains should be avoided. When household cooking was included, its contribution especially to global warming potential of the systems was found to be enormous, with microwave as a positive exception.

Improvement assessment of the dairy farms in cheese production system

Data sets, including feeding inputs, from 700 milk production farms were divided into three categories: farms producing milk at overall average level (7155 kg milk/cow/year), at lowest level (average 6409 kg from 143 farms) and at highest level (average 7906 kg from 144 farms). Increase in milk output is influenced by many inter-related farm parameters. The purpose of the assessment was to form an overview of the importance of the milk output as such. When assessing environmental impacts of different feeding strategies, with different levels of concentrates and crude proteins of concentrates, farm data could not be used because there are many factors influencing milk output and nutrients excretions. Instead, results from 14 milk production trials with 112 different diets were used to study the N and P excretion and washouts with different level of concentrates of the total feeding. Furthermore, results from 27 milk production trials with 186 different diets were used to study the N and P excretion and washouts with different crude protein contents in concentrates (Nousiainen et al. 2003 and Yrjänen et al. 2003). In addition to these contributions of crop yields, nitrogen inputs, lifetime of dairy cattle and density of dairy cattle per ha to environmental impacts of the system were assessed. In studied region, average figures of dairy cows were 3,1 calving times and 2.5 lactation periods. As an example of results, increased number of lactation period decreased rapidly N emissions per cheese ton. This was a result of a dilution of feeds needed to raise a two-year-old heifer and a greater production capacity in later lactations.

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References

Cederberg, C. & Stadig, M. 2003. System Expansion and Allocation in Life Cycle Assessment of Milk and Beef Production. Int J LCA **8** (6) 350-356 (2003).

Katajajuuri, J.-M. & Loikkanen, T. Enhancing competitiveness through green innovation - Finnish food industry committed to environmental data production - LCA pilot case for barley. 9th International Conference of the Greening of Industry Network, Sustainability at the Millennium: Globalization, Competitiveness, and the Public Trust, January 21-25, 2001, Bangkok, Thailand. 9 p.

Katajajuuri & Voutilainen (2002). LCA case studies on Finnish food production chains - management of secondary inflows and outflows. SETAC Europe 10th LCA Case Studies Symposium "Recycling, close-loop economy, secondary resources", 2-3 December, Barcelona, Spain. 6 p.

Katajajuuri, J.-M., Voutilainen, P., Tuhkanen, H.-R. & Honkasalo, N. 2003. Environmental impacts of oat flakes. Maa- ja elintarviketalous 33: 47 p. In Finnish. http://www.mtt.fi/met/pdf/met33.pdf

Nousiainen, J., Kytölä, K., Khalili, H. & Huhtanen, P. 2003. Methods of improving N utilization through feeding strategies on dairy farms. In Uusi-Kämppä, Markku Yli-Halla & Kaarina Grék: Reducing environmental loading from dairy farming. Maa- ja elintarviketalous 25: p. 26-39. In Finnish. http://www.mtt.fi/met/pdf/met25.pdf

Poikkimäki, S & Virtanen, Y. 2003. Supply *web integrated life cycle assessment*. PTR report NO. 49. ISBN 951-8988-30-7.

Virtanen, Y. & Poikkimäki, S. 2003. Applicability of the Product Life-Cycle Analysis. European Brewery Convention, 29th International Congress, Dublin Ireland 2003.

Voutilainen, P., Tuhkanen, H.-R., Katajajuuri, J.-M., Nousiainen & Honkasalo, N. 2003a. Environmental impacts and improvement possibilities of Emmental blue-label cheese. Maa- ja elintarviketalous 35: 91 p. In Finnish. http://www.mtt.fi/met/pdf/met35.pdf

Voutilainen, P., Katajajuuri, J.-M., Tuhkanen, H.-R. & Honkasalo 2003b. Environmental impacts of Kesäpöytä cheese-cream potato gratin and Pirkka potato flour. Maa- ja elintarviketalous 34: 54 p. In Finnish. http://www.mtt.fi/met/pdf/met34.pdf

Yrjänen, S., Nousiainen, J., Kytölä, K., Khalili, H. & Huhtanen, P. The feasibility of minimising faecal excretion on dairy farms. In Uusi-Kämppä, Markku Yli-Halla & Kaarina Grék: Reducing environmental loading from dairy farming. Maa- ja elintarviketalous 25: p. 13-25. In Finnish. http://www.mtt.fi/met/pdf/met25.pdf

Weidema, B. 2001. Avoiding co-product allocation in life-cycle assessment. Journal of Industrial Ecology 4(3): 11-33.

Effects of Intensification of Dairy Farming in New Zealand on Whole-system Resource Use Efficiency and Environmental Emissions

Ledgard, S.F.[1*], Finlayson, J.D.[1], Patterson, M.G.[2], Carran, R.A.[3], and Wedderburn, M.E.[1]
[1]AgResearch Ruakura, Private Bag 3123, Hamilton, NZ
[2]Massey University, Private Bag 11222, Palmerston North, NZ
[3]AgResearch Grasslands, Private Bag 11008, Palmerston North, NZ
*AgResearch Ruakura, Private Bag 3123, Hamilton, New Zealand,
phone ++6478385133, fax ++6478385155, stewart.ledgard@agresearch.co.nz

Abstract
A life cycle assessment (LCA) approach was used to estimate whole system resource use and environmental emissions for the dairy industry in the Waikato region of New Zealand. Application of a "top-down" LCA method using input/output analyses revealed marked differences in the relative contribution between dairy farms, dairy factories and indirect contributors. For example, the relative use of energy by farms : factories : indirect contributors was 1 : 3.0 : 1.8, whereas the corresponding relative greenhouse gas emissions were 1 : 0.2 : 0.3. A "bottom-up" LCA method was used to evaluate the whole farm-system (dairy farm + grazing and forage land) effects of intensification using nitrogen fertiliser or forage crop integration. Fertiliser nitrogen increased production and economic efficiency but decreased environmental efficiency. In contrast, increased use of forage produced off-farm increased the total use of land and production efficiency, with no loss in environmental efficiency (per litre milk).

Introduction
Milk production on dairy farms in New Zealand has been steadily increasing over time. This has occurred through a number of factors including increased feed supply through greater use of nitrogen (N) fertiliser and increased use of supplementary feeds. The effects of this on efficiency of use of resources such as energy, nutrients and land are uncertain.

Evaluation of dairy farm efficiency should go beyond consideration of only the dairy farm unit to incorporate **total** land and use of resources. Ideally, a whole-system evaluation should account for other indirect contributors (e.g. energy used for fertiliser production) and the dairy processing system. As well as resource use efficiency, the whole system evaluation should account for the impacts of intensification on air and water quality. In this paper, LCA methodologies are applied to examine resource use efficiency and environmental emissions in the dairy production chain, and to determine the impacts of intensification practices.

Keywords. Dairy, emissions, intensification, resources, whole-system.

Methodology
Waikato Dairy Industry
A "top-down" LCA approach used Input/Output matrices (Patterson and McDonald, 1996) for 23 or 48 sector models of the Waikato and NZ economies for a range of resource uses and

emissions. These were used to calculate life-cycle multipliers and define the indirect contribution of all industries or sectors to the Waikato dairy industry. This evaluation was based on 1997/98, a recent period when detailed national data was available from StatsNZ. There are about 6000 dairy farms in the Waikato region (about 40% of New Zealand's total) and nine dairy processing factories.

Farm economic, productivity and resource use data for Waikato dairy farms was obtained from dairy industry statistics and from a database (Dexcel ProfitWatch) of 128 farms. The OVERSEER® nutrient budget model (Ledgard et al., 1999) was used to estimate N leaching and total N emissions to waterways from the "average" Waikato dairy farm. It was also used to estimate IPCC-based emissions of the greenhouse gases (GHG), methane and nitrous oxide. Farm emissions of CO_2 were estimated using fuel and electricity data, and emission factors from Wells (2001). Milk input, fuel use and nutrient discharge information for the nine Waikato dairy factories was obtained from the Dairy companies. Energy use data was derived from NZ statistics.

Dairy Farm Intensification

Data for 2000/2001 for the average 83 ha Waikato dairy farm was derived from the Dexcel ProfitWatch database. This farm was estimated to use 15 ha of an average intensive Waikato beef farm for grazing non-lactating animals (grazing replacement animals off farm is common practice in New Zealand). Similarly, the farm purchased forage which would be produced on 2 ha of a 'typical' double-cropping block (yielding 26 t DM/ha/yr of maize and oats silage).

Methodology outlined in the previous section was applied in the evaluation of:

1. Base farm system, with 83 ha dairy farm producing 10 m^3 milk/ha,
2. Base farm system plus extra 200 kg fertiliser-N/ha on the dairy farm producing 12 m^3 milk/ha, and
3. Base farm system plus extra 2 t DM/ha of silage (from an extra 6.4 ha of forage block); assumed to produce 12 m^3 milk/ha on the dairy farm.

All options were assumed to carry 2.8 cows/ha on the dairy farm. Milk production responses were based on average research data. A "bottom-up" LCA method (e.g. Cederberg, 1998) was used for whole-system analysis to the point of milk in the farm vat ready for collection and processing.

Results and Discussion

Resource use and emissions

Relative differences between farms and dairy factories in the use of resources and level of emissions varied greatly (Table 1). For example, the relative use of energy by farms : factories : indirect contributors was 1 : 3.0 : 1.8, whereas the corresponding relative greenhouse gas emissions were 1 : 0.2 : 0.3. There was a large difference in the source of the GHG emissions. Factory emissions were all associated with energy consumption. In contrast, on dairy farms the use of energy only generated about 2% of GHG emissions and the main sources were nitrous oxide and methane from grazing animals. Calculated direct emissions of N to waterways from farms were about 200 times that from factories. Most of the farm N emissions were estimated to come from N leaching losses to groundwater. This evaluation highlighted the benefits of using LCA to determine the relative contributions from different direct and indirect contributors to total resource use and emissions. Thus, it was a valuable tool for identifying inefficiencies in the production system.

Dairy farm intensification

The two intensification options involved increased use of N fertiliser or forage. In terms of land use efficiency, N fertiliser was the most efficient at increasing productivity per unit of land area, particularly when the whole-system land use was accounted for (Table 2). However, for N leaching, the farm emissions per m^3 milk increased by about 70% for the +200N system, and there was a similar increase when estimated on a whole-system basis. In contrast, the forage treatment decreased N leaching per m^3 milk by 10% on the dairy farm, but this efficiency gain was reduced when total land use was considered. Greenhouse gas emissions per m^3 milk were similar for the base farm and +forage system, but increased by about 15% for the +200N system. The latter was mainly due to increased N_2O emissions. This evaluation highlighted that the choice of intensification method influences the potential for gain in dairy farm system efficiency.

Benchmarking

The LCA methodologies were useful for determining “hot-spots” in the dairy production chain, for evaluating the total impacts of land intensification or improved management practices, and for benchmarking farms or whole industries within and between countries. Data for GHG emissions per m^3 milk for the average Waikato dairy farm were similar to those for the Swedish dairy farm of Cederberg (1998). While the energy-related CO_2 emissions were greater for the Swedish farm, this was countered by lower methane emissions per unit of milk from high-producing Swedish cows. Total farm energy use per unit of milk production on the Swedish farm was over 5-fold higher than that of the Waikato farm on a whole-system basis. This was mainly due to high fuel use in the Swedish farm system for crop production, feeding and heating the farm dairy. The NZ farm system with all-year-round grazing of long-term permanent legume-based pastures is a low energy requiring system, but this advantage may diminish with intensification. Further research is required to determine whether the energy advantage of the NZ farm system is sufficient to compensate for “food-miles” or the energy cost associated with shipping dairy produce from NZ to Europe.

References

Cederberg, C., 1998. Life cycle assessment of milk production – a comparison of conventional and organic farming. *SIK report*. The Swedish Institute for Food and Biotechnology. SIK, Gothenburg, Sweden.

Ledgard, S.F., Williams, P.H., Broom, F.D., Thorrold, B.S., Wheeler, D.M. and Willis, V.J., 1999. OVERSEER™ – a nutrient budgeting model for pastoral farming, wheat, potatoes, apples and kiwifruit. In: Currie, L.D., Hedley, M.J., Horne, D.J. & P Loganathan, P. (eds.). *Best soil management practices for production.* Massey University, NZ. pp 143-152.
Patterson, M. G. and McDonald, G., 1996. Regional level environmental accounting systems in New Zealand, using Input Output methodologies. In: *Tracking progress: Linking environmental economy through indicators and accounting systems*. Australian Academy of Science, Fenner Conference, The University of New South Wales, Sydney.
Wells, C., 2001. Total energy indicators of agricultural sustainability: Dairy farming case study. *Report to MAF; Technical Paper 2001/3*. 79pp.

Table 1. Some examples of direct resource use and emissions from Waikato dairy farms and factories, the indirect contributions and total embodied values for the whole Waikato dairy industry.

	Farms (direct)	**Factories (direct)**	**Indirect contributors**	**Total**
Energy use (TJ)	997	2970	1843	5810
GHG (Gg CO_2-equiv.)	3074	616	1030	4720
N emission to water (Gg)	19.4	0.1	7.0	26.5

Table 2. Effect of intensification from 10 (average) to 12 m^3 milk/ha, using N fertiliser (+200 kg N/ha/yr) or extra bought in forage (+2 t DM/ha/yr as maize + oats silage), on environmental emissions. Data for the dairy farm only and the whole-system (dairy farm + grazing + forage land) are compared.

	Dairy Farm			**Whole-system**		
	Av.	**+200N**	**+Forage**	**Av.**	**+200N**	**+Forage**
Milk (m^3/ha/yr)	10.0	12.0	12.0	8.3	10.0	9.4
N leached (kg/ha/yr)	36	74	38	32	64	35
GHG (kg CO_2-equiv/ha/yr)	8590	11970	10780*	7790	10590	9940*
Efficiency indices:						
kg N leached/m^3 milk	3.6	6.2	3.2	3.9	6.4	3.7
kg CO_2-equiv/m^3 milk	859	998	898*	939	1059	1058*

* does not account for CO_2 emissions from cultivated soil

Food purchasing processes and environmental information in the food service industry in Sweden

Bergström, K., Solér, C., Adolfsson, A., Shanahan, H.

** Kerstin Bergström, PhD student, Göteborg University, Department of Home Economics, Box 300, 405 30 Göteborg, Sweden, Phone +46 (0)31-773 4204. Fax +46 (0)31-773 4209, Kerstin.Bergstrom@ped.gu.se*

Cecilia Solér, PhD, Gothenburg School of Economics and commercial Law, Department of Business Administration, Box 600, 405 30 Göteborg, Sweden, Cecilia.Soler@cfk.gu.se

Anna Adolfsson, PhD student, Göteborg University, Department of Home Economics, Box 300, 405 30 Göteborg, Sweden, Anna.Adolfsson@ped.gu.se

Helena Shanahan, ass. professor, Göteborg University, Department of Home Economics, Box 300, 405 30 Göteborg, Sweden, Helena.Shanahan@ped.gu.se

Abstract

Food consumption affects the environment in several ways. In Sweden one fifth of all food is consumed outside homes. It is therefore highly justified to study food purchasing processes and needs for environmental information in the food service industry. Purchasing managers have been interviewed in food production companies, wholesalers, local and regional public authorities, restaurant catering and retailing. They actively apply quality assurance in order to reduce environmental impact of cooling media, transport and packaging. However purchasing managers lack knowledge about environmental impact of other stages in the food chain. This makes it difficult to value impacts of foods and to make environmentally sound purchase decisions.

Keywords: environmentally friendly foods, procurement, institutional and restaurant catering.

Background

Today's food consumption affects the environment in numerous ways. Throughout the life cycle of food, which includes agricultural production, storage, transportation, processing, preparation and waste disposal, resources are used and emissions are released to the environment. This project has a focus with substantial potential for broadening the understanding of how purchasing managers could contribute to a more environmentally friendly food industry as well as suggesting templates for tailoring the information required by users.

The research programme is titled: *Designing and evaluating the impacts of environmental information in food service institutions and the food wholesale sector.* The project is supported by the Foundation for Strategic Environmental Research, MISTRA, 2002-2004. It focuses on how different factors interact in food purchasing, especially how environmental information related to foods affects food purchase in institutional catering and the wholesalers sector. The project is a co-operation between Göteborg University, the Mid Sweden University, Swedish University of Agricultural Sciences, the Swedish Defence Research Agency and the University of Trollhättan/Uddevalla. In the initial phase of the project, implemented in the autumn

2002, purchasing managers in the commercial and public companies were interviewed. In the phase to follow the environmental impact of selected foods will be calculated using a life-cycle inventory approach. Based on these calculations different types of environmental information labeling will be developed and tested on purchasing managers in the commercial and public sectors. The effects of different environmental purchase decisions based on labeling will be studied and illustrated in a number of scenarios describing changes in resource use in the food service industry. The long-term objective of the project is to contribute to patterns of food production and consumption with substantially lower resource use and emission levels than today. This will contribute to societal goals to safeguard ecosystems for sustained generation of ecological services, which is a mutual prerequisite for economic and societal sustainability.

Purpose of the poster

The purpose of the poster is to report findings from the initial phase which studied practices and needs in relation to environmental information of producers, suppliers and corporate customers within the food service industry.

Method used

Informal interviews were held with purchasing managers in national, regional and local authorities in food service institutions, and with restaurant and retailing managers as well as managers as suppliers of food. The interviews focused on decision situations as well as organisational specific factors that affect the use of and perceived need for environmental information. The interviews were phenomenological. This method focuses on agents personal views on certain subjects and results in descriptions of the agents experience. Here the description of experiences focused on the work of using and communicating environmental information.

Results

The results point out that purchasing is a complex information situation. The procurement is an important and time consuming work for purchasing managers. Communication between participating companies, departments and suppliers in the study is illustrated in table 1. The information about inquiries, procurement, agreements, purchasing, deliveries and other food items flows in many directions.

The phenomenological analysis shows that the respondents represent four different perspectives when using environmental information in the purchasing process. The first perspective can be called “to regard financial facts”, the second is “to work according to the law”, the third is “to adjust according to demands” and lastly “to be in control”.

All the respondents said that environmental information about food is needed, when making inquiries about food for procurement and when negotiating written agreements. The information is also needed in training programmes for employees, in production, and when labeling and marketing. It is also needed when making quality revisions.

Conclusion

It can be concluded that the purchasing process is very complex and that information and food items, at different degree of processing, flow in many directions. Moreover, purchasers have different perspectives on environmental purchase decisions. In both the commercial and public food service industry, work is actively carried out through quality assurance to reduce the environmental impacts of foods. Areas included especially are cooling media, transport and packaging. However, purchasing managers lack knowledge about environmental impacts from other stages in the food chain. This makes it difficult to value the total impacts of foods on the environment and consequently to make environmentally sound purchase decisions.

Table 1. The table is illustrating the participating companies, departments and suppliers in the study and the connections between them.

	Food producers	**Food suppliers/ Whole sales**	**Public food service in local and regional authorities**	**Commercial and public food service through catering chains**	**Retailing**
Food purchase	Food purchasing process for producers, also being suppliers	Food purchasing process for wholesalers, being suppliers	Food purchasing process for public catering in local government agencies and county councils	Food purchasing process of food for global catering chains	Food purchasing process for retailers
Food production			In-house food production in canteens with public management	Food production in catering for canteens with contract management	Food sales in retailing stores. Food preparation in private households
Food consumption			Meal consumption in canteens and dining rooms	Meal consumption in canteens and dining rooms	Meal consumption in private households

eLCA: website and database of IPP tools for SMEs in the agro-food sector

Milà i Canals, L., Fullana, P., Mantoux, F.

* Escola Superior de Comerç Internacional, Pg. Pujades 1, ES-08003, Barcelona (Catalonia), Spain. +34 93 415 11 46. +34 93 295 47 20. llorenc.mila@admi.esci.es

Abstract

The new approach of environmental policy in Europe, the so-called Integrated Product Policy (IPP), seeks to improve environmental performance of products and services in a life cycle perspective by integrating different tools of reduction of environmental impacts. Examples of such tools are LCA (Life Cycle Assessment) and eco-design, eco-labelling, green purchasing and ecotaxes. The major barriers for its incorporation by Small- and Medium-sized Enterprises (SMEs) concern the data availability as well as the knowledge and resources to manage this information. The European project eLCA aims to support SMEs in the implementation of IPP tools by providing through a web portal all the technical and managerial information (pre-elaborated LCA data, guidelines for the integration of LCA and eco-design, legislation, description of market tools, etc.), training resources and case studies, specialised software (LCA, eco-design) and on-line consulting services. Thus, the SMEs of the agro-food sector will find Good Agricultural Practices (GAP), simplified LCA data for the conduction of LCA, sources of economic incentives from the CAP reform, etc.

Keywords: IPP, tools, LCA extension, Internet.

Introduction: the eLCA project

Integrated Product Policy (IPP) is a new approach for environmental policy in Europe. It aims to improve the overall environmental performance of products and services by integrating different environmental tools and stakeholders. European policy in this area includes LCA, and its application to eco-design, eco-labelling and "green" purchasing, as evidenced in the new EC communication of June 2003. Small- and Medium-sized Enterprises (SMEs) provide the majority of economic activity across Europe but are unlikely to implement IPP because of the high cost of gathering data as well as the knowledge and resources to manage this information.

eLCA is a pan-European project funded by the Commission under its eContent programme. It provides a web portal (ecosmes.net) containing pre-compiled life cycle data linked to a simplified life cycle analysis tool in order to overcome the barriers that prevent SMEs from using LCA. The web site also provides information on eco-design, legislation, market tools, as well as training packages and guidelines derived from case studies. The database will be pre-populated with the results of several product chain studies. Stakeholders will use the portal to carry out studies on their product, supplying data at the same time as using the information. This use of Internet will benefit the environment by facilitating rapid access to environmental information and tools for SMEs allowing them to respond to "green" procurement and other supply chain pressures to reduce the environmental impacts of their products.

The overall objective of the web portal is to develop the market for IPP services, creating a platform for consulting services on training and LCA-based studies and tools. Partnership agreements are envisaged for the distribution of consulting services in many sectors. The partners of the project include public agencies, research centres, Universities, trade associations and service centres, and consultancies.

Demand for environmental information in the agro-industrial sector

SMEs represent also the majority of industries in the agro-industrial sector, and are generally run by small farmers who have not the knowledge to incorporate all the environmental issues at stake in their decision-making. It has been suggested that the farmer's decisions on technique implementation may have a greater effect on the environmental performance of the agricultural system than the choice of technology (Milà i Canals, 2003). Other factors such as the physical site conditions (soil type, weather, etc. see Cowell and Clift, 1998, and Milà i Canals, 2003), and the socio-economical context of the farm affect the environmental performance of agricultural systems (see Figure 1).

The socio-economical context in the agro-industrial sector is formed by a wide range of actors, from the inputs suppliers (fertilisers, pesticides, seeds…) to the final consumers. The IPP approach actually pretends to integrate all the information and communication tools that work between these actors, in order to foster a reduction of the environmental impacts of the whole supply chain, instead of focusing on the production stage. This integration includes legislative pressure but also communication and information exchange between the actors in the supply chain, e.g.: retailers asking for environmental performance to their suppliers, consumers demanding eco-labelling, etc.

In this context, agro-industrial SMEs are joining efforts with research centres and public bodies in several research projects aiming at facilitating adoption of environmental tools. These projects have focused on process-based tools such as Environmental Management Systems (EMS) and recommendations on Good Agricultural Practices. The IPP approach is an opportunity to expand the research topics to product-focused tools such as LCA.

Wholesale retailers have been active for a long time in response to consumers' demand of information, and have set their own programmes for product information (type-I-like eco-labelling schemes). These programmes, together with the increasing presence of EMS within the sector, are increasing the demand for information upstream in the product chain. The increased use of LCA will facilitate the provision of this information.

Finally, the consumers' pressure and demand for environmentally friendly agrarian products is another driver for the multiplication of environmental product information systems requiring for life cycle data.

eLCA and the agro-industrial sector

The eLCA project is presenting the first results for several industrial sectors in September 2003, but the agro-industrial sector is not yet included. So far, the activities of the project in this sector are focused on the **dissemination of LCA**. It is essential that this dissemination be done so the farmers perceive LCA as a way to increase their competitiveness, to facilitate legislation compliance or the communication with the rest of the supply chain, in order to get positive reactions.

In Spain, where the extension of the eLCA services to the agro-industrial sector is first planned, the activities will begin with LCA training to relevant stakeholders: research and training centres, Ministry for agriculture… This training is aiming at fostering the demand for LCA studies and LCA data. Then, product panels will be organised in strategic sectors, such as fresh fruit and pork. Product panels serve as a start for supply chain studies and definition of Good Agricultural Practices (GAP), which in turn provide the basic information for the pre-population of the website. Apart from simplified LCA data for the conduction of LCA, the SMEs of the agro-food sector will find training and consultancy on GAP, sources of economic incentives from the CAP reform, tools to communicate with the other actors of the supply chain, etc. Partnership agreements with stakeholders of the agro-industrial supply chain seek the promotion of the use of the website, and include the integration of already existing services and contents.

References

Cowell, S.J. and Clift, R. 1998. Site-dependency in LCAs involving agricultural production. *8th Annual Meeting of SETAC-Europe. Abstracts*. SETAC. 14-18 April 1998. Bordeaux, France.

Milà i Canals, L. 2003. Contributions to LCA Methodology for Agricultural Systems, Site-dependency and soil degradation impact assessment. *PhD Thesis*. Universitat Autònoma de Barcelona, Spain.

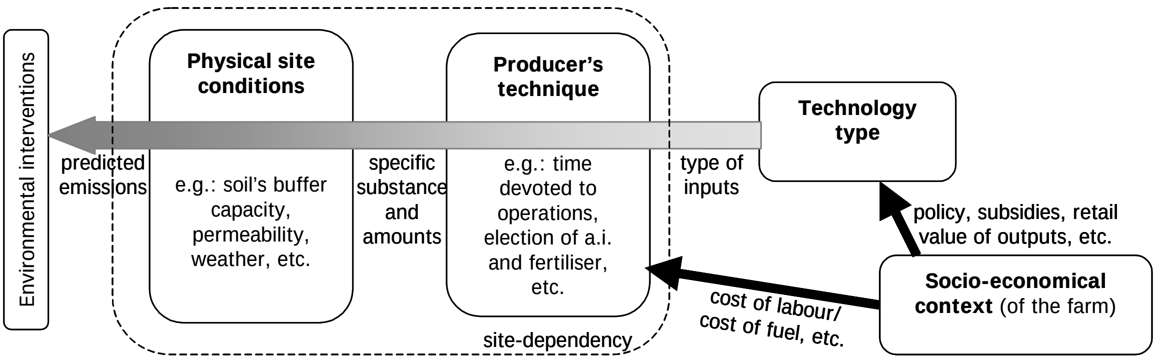

Figure 1. Aspects influencing the environmental impacts of the farm (Milà i Canals, 2003).

Sources of Site-Dependency and Importance of Energy Consumption in Agricultural LCA: Apple Production in New Zealand.

Milà i Canals, L., Burnip, G.M., Suckling, D.M. and Cowell, S.J.

** Escola Superior de Comerç Internacional, Pg. Pujades 1, ES-08003, Barcelona (Catalonia), Spain. +34 93 415 11 46. +34 93 295 47 20. llorenc.mila@admi.esci.es*

The Horticulture and Food Research Institute of New Zealand Ltd, PO Box 51, Lincoln (Canterbury), New Zealand

The Horticulture and Food Research Institute of New Zealand Ltd, PO Box 51, Lincoln (Canterbury), New Zealand

Centre for Environmental Strategy, The University of Surrey, Guildford (Surrey), GU2 5XH United Kingdom

Abstract

Research conducted in New Zealand apple orchards shows that farmers' technique exerts a considerable effect on the LCA results, introducing variances of e.g. 30-50% in energy consumption when different farmers perform the same field operation. Energy consumption is found to be significantly higher in organic farming than in integrated farming in apple production in New Zealand, and it contributes above 50% to most impact categories considered in the study. Therefore, holistic approaches such as LCA covering the different environmental impacts from agriculture should be promoted when designing certification schemes or assessing the environmental soundness of agricultural technologies.

Keywords: apple production, site-dependency, farmer technique, New Zealand, comparative LCA

Introduction: Goals and scope of the study

The LCA study aims at detecting the environmental hotspots of two different systems for apple production, Integrated Fruit Production (IFP) and Organic Fruit Production (OFP), in two New Zealand regions: Central Otago (CO) and Hawke's Bay (HB) (Milà i Canals et al., 2001). This presentation analyses the effect of farmer's technique on the results using data from specific sites (Milà i Canals, 2003).

The functional unit has been set to the production of 1 ton of Braeburn apples of export or local market quality in New Zealand. The system's physical boundaries are set in the whole orchard, including a tree wind shelter that is usually found in New Zealand apple orchards. On the vertical axis, soil is considered as part of the system (and thus of technosphere) down to a depth of 1 m. Substances crossing these boundaries will be considered as emissions to the environment. In the case of ancillaries, only machinery use has been analysed. Soil quality degradation has not been assessed due to lack of methodology. Farming infrastructure (buildings, irrigation infrastructures, etc.) and its maintenance has neither been included. As for the time boundaries, only one year of the orchard's high yield period has been considered in the study. Even though soil quality has not been assessed in the apple LCA, the substances emitted to soil remaining in soil after the time boundaries are crossed (at harvest) are also considered as an emission to soil. Their inclusion is consistent with the need of leaving the soil in the same conditions as it is found in the beginning of the system (Audsley et al., 1997, p. 85). Finally, from a life cycle perspective, only the phases from cradle to gate are analysed, as the transportation of the finished product, consumption, and final waste disposal are not relevant for the purposes of the study.

Data for agricultural inputs consumption and agricultural practices were obtained directly from four individual producers (one for each technology and region), who filled a questionnaire for the season 1999-2000. Some further checking had to be done by telephone interviews in most cases from June to August 2000.

The impact assessment phase includes the impact categories usually considered in LCA: global warming, photochemical oxidants formation, acidification, nutrification, human toxicity (air, water and soil), ecological toxicity (acute and chronic for aquatic ecosystems and chronic for terrestrial ones) (Milà i Canals et al., 2001). Characterisation factors were obtained from Hauschild and Wenzel (1998), and new toxicity factors have been calculated for the pesticides used in IFP using the method described in Hauschild and Wenzel (1998) (see Milà i Canals, 2003). Besides, an indicator on the competition aspect of land use has been included as a measure of land use efficiency, expressed in ha year. Finally, a simple indicator expressing the amount of non-renewable energy consumed in each site is included.

The inventory parameters mostly determining the life cycle impact assessment (LCIA) results (dominance analysis) are checked for their confidence by estimating error margins. This qualitative uncertainty analysis assures the consistency of LCA results.

System's description and LCI

The production operations have been deeply analysed as part of the foreground system for the apple LCA, including detailed calculations for field emissions and energy consumption for mechanisation. Data for the processes in the "background system" (agro-chemicals production, fertilisers production, machinery production and delivery of energy carriers and transportation) was gathered from the literature, with the exception of the substances for the biological pest control used in OFP, which have been described in this study (Milà i Canals, 2003).

Field processes

Figure 1 is a graphical representation of the operations taking place during apple production, with the direct inputs to the apple orchard and a rough graphical representation of the timing of different operations for New Zealand, from May to July (when the trees are pruned) to April (harvest ends). Field emissions have been considered mainly for understorey management (herbicide emissions in IFP), fertiliser use, thinning (only in IFP as well), and pest and disease management. Energy consumption, on the other hand, has been studied for all field operations. Table 1 depicts the operations in which differences between regions and technologies exist.

Field emissions

Emissions of ammonia, nitrate, nitrous and nitrogen oxides, methane and heavy metals are estimated for the fertilisers used from references in the literature. In the case of synthetic pesticides, a detailed partition analysis is done based on Hauschild (2000) to estimate emissions to air, surface water, groundwater and soil. Heavy metals emissions are also considered for mineral pesticides.

Results

Figure 2 gives an overview of the relative contribution to each impact category by the sites under study. It must be noted that IFP systems have lower values for land competition than OFP ones, i.e.: they produce more apples in less surface, even though OFP_HB has a similar value for land competition than IFP orchards. The land competition factor has an obvious effect on the LCA results, as systems with lower productivity and pack-out will be charged more for their impacts. Energy consumption determines the impacts on photochemical oxidants formation and ecological toxicity, and to a lesser extent the human toxicity through air emissions, acidification and global warming (the two latter also affected by fertiliser emissions in IFP), and this is why a further analysis of energy consumption is given in Figure 3.

In the case of human toxicity through water and soil emissions, only integrated sites show relevant impacts (see Figure 2). These impact categories are dominated by emissions of synthetic pesticides, which are only used in IFP systems. The huge differences (above one order of magnitude) between the IFP sites are due to local conditions (mainly the soil type) and to farmer's practices (choice of active ingredient, method of application etc.). Finally, ecological toxicity in aquatic ecosystems is dominated by emissions related to direct energy consumption and to inherent energy used in the production of inputs (machinery, pesticides and fertilisers), and no clear differences may be observed between organic and integrated systems.

Sources of energy consumption

Figure 3 shows the systems consuming more energy, and the sources of energy consumption. What first comes into sight from the figure is that organic systems have higher energy consumption than IFP systems. Of all the producers participating in the study, OFP_HB is the one with the highest energy consumption, due to the extraordinary use of hydra-ladders for pruning, thinning, and harvesting. The energy consumption for understorey management in OFP_CO is also noticeable, and can be explained by the intensive mowing of the orchard; also the fact that the engine of the tractor is of a higher cc rating than usual partly explains this higher contribution.

From Figure 3 it is obvious that direct energy consumption by field operations is the main cause of energy consumption (70-75% in IFP and 83-90% in OFP). Pruning and thinning have a greater share in organic systems than in integrated ones, due to the higher mechanisation needed to perform these operations manually. The contribution of pesticides production to energy consumption is noteworthy in integrated systems, where it represents from 11% to 18%. Also energy consumption related to machinery production is relevant, and contributes 7% to 15% to total consumption.

Discussion

The first conclusion arising from these results is that the results are highly dependent on the characteristics of the site. Firstly, most impacts are directly dominated by producer's practices in some way or other: selection of fertiliser or pesticide active ingredients, efficiency in the use of machinery, etc. For instance, the same field operation (e.g.: mowing, thinning, pruning, harvesting…) performed by different farmers results in variances of 30-50% in energy consumption. Physical site conditions (particularly soil type) also have a significant effect on the impact categories, mainly through their effect on field emissions; they act as "filters" reducing or increasing emissions to the environment from the amounts used by the farmer.

Besides, it can be stated that integrated production presents a wider variety of impact sources than OFP, and these are both related to energy emissions and field emissions from pesticides and fertilisers. In the case of organic apple production, energy consumption is a clear focus of impact generation. This is because inputs used in organic fruit production are in principle less problematic than those used in IFP. Apart from this overall distribution of impact sources, ample variations appear in the relative contributions of each producer's item to the impact categories.

In summary, site characteristics have been found to affect the LCA results to a bigger extent than the choice of technology (organic or integrated) in many impact categories. Actually, only the impact categories that are clearly affected by substances only used in IFP (synthetic pesticides) show clear differences between IFP and OFP (Human Toxicity through water and soil). Also those impact categories chiefly dominated by energy consumption (Eco-Toxicity through soil and Photochemical Oxidants Formation) present clear differences, because in the New Zealand apple LCA consistently higher energy consumption has been found for organic orchards.

Energy consumption (mainly related to the intensive mechanisation of field operations) seriously hampers the environmental preference of OFP over IFP in New Zealand, where it is significantly higher in organic farming than in integrated farming. Above 50% of most impact categories considered in the study is due to energy-related emissions. The degree of mechanisation should thus be considered when designing certification schemes or assessing the environmental soundness of agriculture.

References

Audsley, E. (coord.), Alber, S., Clift, R., Cowell, S., Crettaz, P., Gaillard, G., Hausheer, J., Jolliet, O., Kleijn, R., Mortensen, B., Pearce, D., Roger, E., Teulon, H., Weidema, B., Van Zeijts, H. 1997. Harmonisation of Environmental Life Cycle Assessment for Agriculture. Final Report. Concerted Action AIR3-CT94-2028. European Commission. DG VI Agriculture.

Hauschild, M. and Wenzel, H. 1998. Environmental Assessment of Products. Volume 2 Scientific background. Chapman & Hall, London, UK.

Hauschild, M.Z. 2000. Estimating pesticide emissions for LCA of agricultural products. B.P. Weidema and M.J.G. Meeusen (eds.). *Agricultural data for life cycle assessments. Volume 2*, LCANet Food, pp. 64-79. The Hague, The Netherlands.

Milà i Canals, L., Burnip, G.M., Suckling, D.M., Cowell, S.J. 2001. Comparative LCA of apple production in New Zealand under two production schemes: Integrated and Organic Fruit Production. *International Conference on LCA in Foods. Proceedings*. Gothenburg, Sweden, 27th April 2001.

Milà i Canals, L. 2003. Contributions to LCA Methodology for Agricultural Systems, Site-dependency and soil degradation impact assessment. *PhD Thesis*. Universitat Autònoma de Barcelona, Spain.

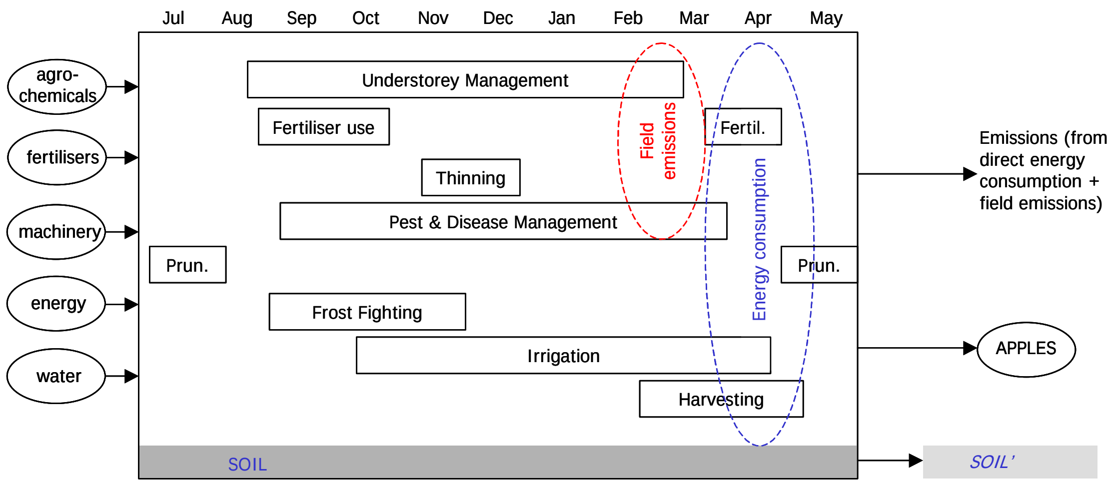

Figure 1. Field operations in the production stage of the apple life cycle in New Zealand (Milà i Canals, 2003).

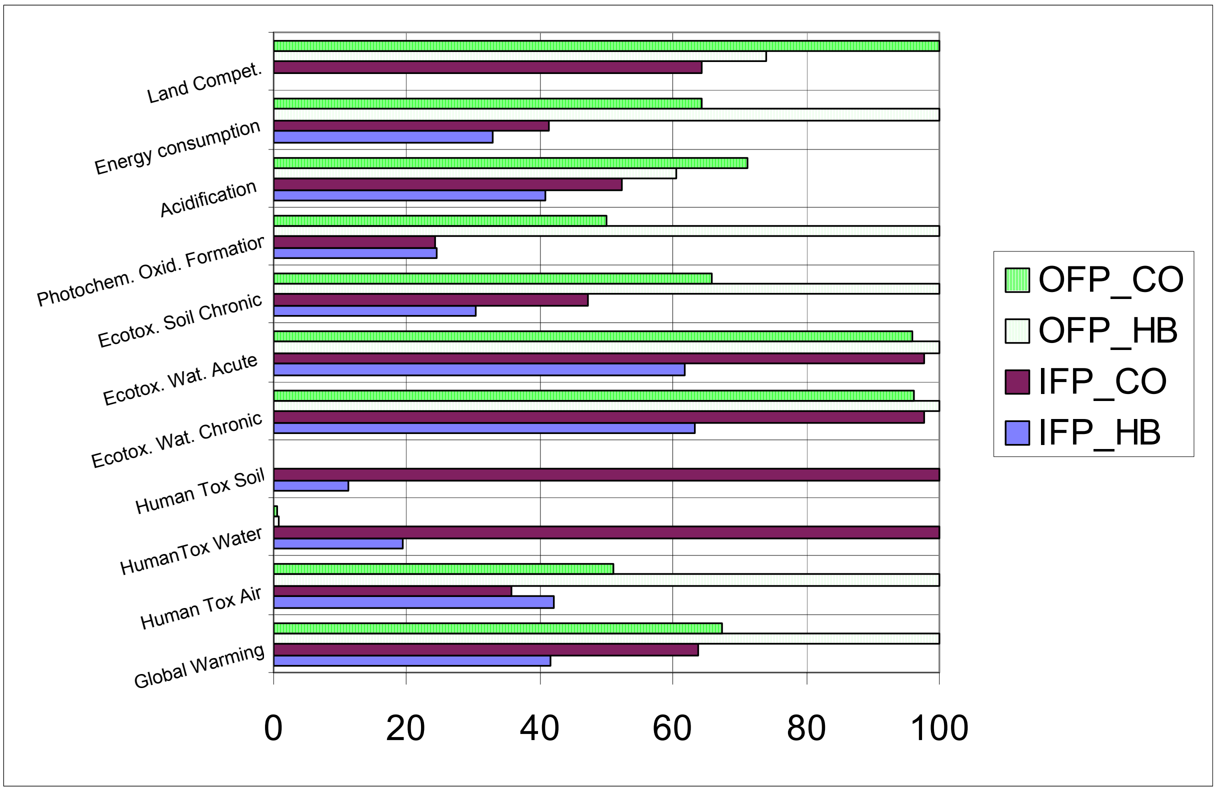

Figure 2. Main results of the New Zealand Apple LCA.

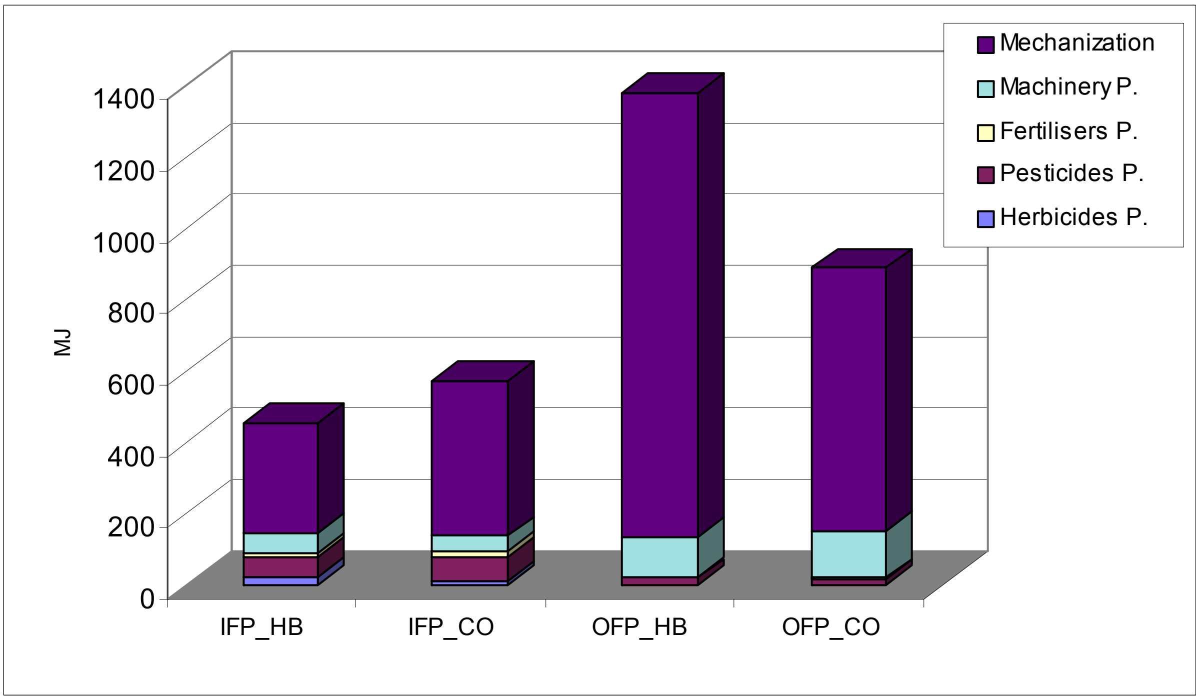

Figure 3. Energy consumption by different input items. Results refer to 1 ton of apples of export or local quality.

Table 1. Main differences in field operations as depending from technology and region (Milà i Canals, 2003).

Operation	***Differences in technology type (IFP / OFP)***	***Differences in regions***
Understorey Management	YES (use of herbicides in IFP; mulching more usual in OFP)	NO
Fertilising	YES (type of fertilisers)	NO
Pest and Disease Management	YES (approach to pest management and type of substances)	YES (intensity of pests related to climate)
Pruning	YES (more time-intensive in OFP; fate of prunings)	NO
Thinning	YES (chemical thinning in IFP; type of substances)	NO
Irrigation	YES (slightly higher water consumption in OFP expected)	YES (source of water; irrigation system)
Frost Fighting	NO	YES (greater need in CO than in HB)
Harvesting	NO	NO

Procedures to improve scope definition and inventory analysis in LCAs of farming systems

Hayo M. G. van der Werf and Claudine Basset-Mens

*INRA, UMR Sol Agronomie Spatialisation, 65, rue de Saint Brieuc CS 84215, 35 042 Rennes Cedex, France. Email :*derwerf@roazhon.inra.fr

We identify three methodological weaknesses in Life Cycle Assessments of farming systems.

- Definition of the technology used is often based on data for a single farm. Compared with other economic activities agriculture involves many production units (farms) with major variability in production processes.
- Data on economic and environmental outputs often poorly represent the processes concerned with respect to specific farmer practices and the effect of climate.
- No assessment of the uncertainty of results is made.

We propose procedures to better address these topics.

This study compared three scenarios for pig production: a) Good Agricultural Practice (GAP), b) quality label "Label Rouge" (LR) and c) "Agriculture Biologique" (AB).

"Technology coverage", i.e. the determination of the technology used, is part of the scope definition. The technology used was defined for crop production, feed production, housing type, access to pasture, storage, treatment and field application of manure. It was preferably based on the official requirements and the production rules for GAP, AB and LR. Other sources used were, in order of preference: statistical data, literature references, data from industry and expert opinions. If these were not sufficient, farms were visited to collect the data. Generally data sources were of better quality for GAP than for AB, with LR intermediate. This approach allowed us to determine technology coverage for each scenario with a satisfactory degree of confidence.

Data quality has a major influence on results. In agriculture both economic and environmental outputs of processes are strongly affected by farmer production practices (e.g. timing of fertilisation) and climate factors. Output data should be as specific and representative as possible, taking into account farmer practices and climate. Output data were preferably based on simulation models taking into account technology, practices and climate. If an appropriate model was not available, we used measured data collected under conditions corresponding closely to those of our scenarios. If fully representative models or data were not available, we used less specific models or data, as published in the international literature. As a last resort, we relied on expert opinion.

In order to assess uncertainty of results we first identified key parameters of economic and environmental output. For each of these a high and low value was defined in addition to the default reference value. High and low values were chosen to reflect realistic rather than extreme values, so that the uncertainty interval defined by these values would contain 60% to 70% of the variability for the parameter concerned. Values corresponding to improved economic output or lower emissions were labelled “favourable”, conversely, values reflecting worse economic output and higher emissions were labelled “unfavourable”. By combining on the one hand all “favourable” values for key-parameters in a “favourable” scenario and on the other hand all “unfavourable” values in an “unfavourable” scenario we obtained two sub-scenarios (“favourable” and “unfavourable”) for each farming system, reflecting an assessment of overall uncertainty.

Predicting the future: comparing pork with Novel Protein Foods on environmental sustainability impairment

*Helms, M.**

**Institute for Environmental Studies, Vrije Universiteit, De Boelelaan 1087, 1081 HV Amsterdam, The Netherlands, Tel: +31 (0)20 444 9557, Fax: +31 (0)20 444 9553, Email: martine.helms@ivm.vu.nl*

Abstract

A method is being developed to assess the environmental sustainability of protein production chains and facilitate a deliberate consumer choice. Pork is compared to a virtual plant protein product based on green peas. Application of the method to the latter product requires an abstract approach in which sustainability is assessed on the basis of impaired ecosystem functioning by a representative selection of production processes. The present focus is on primary production and the regulatory parameters selected are biodiversity, carbon, nitrogen and water.

Introduction

The present manner in which food is produced and consumed has large impacts on the environment. These impacts are expected to increase due to a growing world population and increasing consumption of animal products. As a result, the sustainability of future food production systems is questioned (e.g. see (Tilman et al., 2002). One option to change the system is on the consumer side in the choice of the preferred diet and the selection of foodstuffs.

PROFETAS (PROtein Foods, Environment, Technology And Society) is a research program concerned with the development of more sustainable food production systems (see (PROFETAS, 2002). The focus is on protein production and consumption and the prospects to replace meat in the Western diet with new plant protein products – the so-called Novel Protein Foods (NPFs). The primary hypothesis is that a substantial shift from animal to plant protein foods is a) environmentally more sustainable than present trends, b) technologically feasible, and c) socially desirable.

The present study focuses on the first part of the hypothesis and examines the development of a method to assess the environmental sustainability of protein production chains. This should facilitate the consumers' choice between the more sustainable of alternative foodstuffs. A program-wide case study has been adopted in which the conventional pork production system is compared to the production of a Novel Protein Food based on dry green peas.

Environmental sustainability assessment

An interesting complication is the virtual nature of the NPF-chain, which precludes the use of conventional LCA: the product has yet to be developed, hence the details of the production system are largely unknown. The potential applications of the method to products with unavailable data, plus a general desire to reduce LCA-like data requirements, argue for a less detailed approach. Therefore, a method is proposed in which the "less detailed approach" is sought in reduction of the complicated and elaborate production chains to a small number of essential processes. These are selected on a macrolevel to make them representative of agricultural based food production in general and are than applied to the pork-NPF example (table 1).

The environmental performance of the main processes is analysed as a measure of impaired environmental sustainability. The latter is considered an anthropocentric concept in which human welfare and permanence of human society is a central issue. This means that the value and importance of the environment is strongly tied to the generation of natural capital that can be used by humans and the permanence of ecosystem services, which provide natural capital and essential living conditions. De Groot et al. (2002) provide a long list of ecosystem services classified in four groups: *regulation functions* such as climate regulation, soil formation and pollination, *habitat functions* – the provision of refugia and nurseries -, *production functions* such as primary production and genetic resources, and *information functions* for spiritual, educational and recreational uses. A long-lasting generation of these ecosystem services requires that a minimum level of ecosystem health and functioning be retained.

The proces of parameter selection

Crucial to the method of assessment is the selection of parameters describing the impairment of ecosystem health and functioning. Focus is on the translation of resource use and environmental problems to the mechanisms behind the loss of regulation, habitat and production services.

The main environmental problems of, here for example, agricultural activities in the primary production phase, are listed in table 2. The importance of specific resource uses or environmental impacts are determined by the contribution of the process(es) to total human-driven impact or human use. In general, human dominance of the global ecosystem is most prominent in land transformation, marine ecosystems, the C-cycle, the hydrological cycle, the N-cycle, the synthesis of persistent organic chemicals and in changes in biodiversity (Vitousek et al., 1997). In turn, the latter is mainly determined by changes in land use, CO2-concentration in the atmosphere, nitrogen deposition and acid rain, climate change and the introduction of exotic species (Sala et al., 2002). Returning to the example of agriculture; it plays a prominent role in land conversion and domination of the C-cycle, the N-cycle and the hydrological cycle (Helms and Aiking, 2003).

Currently the translation to sustained ecosystem functioning must be made and it is these cycles (carbon, nitrogen, water) that may play an important role. Alexander et al. (1997) tell us "Managing and finding solutions to many of the important environmental problems facing humanity begin with understanding and integrating biogeochemical cycles and the scales at which they operate". Tilman (1997) confirms the importance of closing cycles – "Ecosystems attain a sustainable level of functioning when (…) rates of loss and gain of organic matter and nutrients are in balance" – and emphasises the role of species richness in this matter.

Conclusion

Although presently finer details still lay hidden, the importance of the complex interactions between biogeochemical cycles, biodiversity and ecosystem functioning as well as the degenerating effect of food production is clear. Therefore, the pork and NPF chains should be compared primarily in relation to impaired regulatory function of biodiversity, the C-cycle, the N-cycle and the hydrological cycle.

Aknowledgement

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Literature Cited

De Groot, R. S., M. A. Wilson, and R. M. J. Boumans. 2002. A typology for the classification, description and valuation of ecosystem functions, goods and services. *Ecological Economics* 41:393-408.

Helms, M. and H. Aiking. 2003. Food and the environment: towards sustainability indicators for protein production. Proceedings of ECOSUD 2003, Fourth International Conference on Ecosystems and Sustainable Development, June 4-6, Siena.

PROFETAS. 2002. "PROFETAS - Protein Foods, Environment, Technology and Society." Available from http://www.profetas.nl.

Sala, O. E., F. S. Chapin III, J. J. Armesto, E. Berlow, J. Bloomfield, R. Dirzo, E. Huber-Sanwald, L. F. Huenneke, R. B. Jackson, A. Kinzig, R. Leemans, D. M. Lodge, H. A. Mooney, M. Oesterheld, N. LeRoy Poff, M. T. Sykes, B. H. Walker, M. Walker, and D. H. Wall. 2002. Global biodiversity scenarios for the year 2100. *Science* 287, no. 10 March 2000:1770-1774.

Tilman, D. 1997. Biodiversity and ecosystem functioning. In *Nature's services. Societal dependence on natural ecosystems*, edited by Daily, G. C. (Washington DC: Island Press).

Tilman, D., K. G. Cassman, P. A. Matson, R. Naylor, and S. Polasky. 2002. Agricultural sustainability and intensive production practices. *Nature* 418, no. 8 august 2002:671-677.

Vitousek, P. M., H. A. Mooney, J. Lubchenco, and J. M. Melillo. 1997. Human domination of Earth's ecosystems. *Science* 277, no. 25 July 1997:494-499.

Table 1. Main processes in pork and NPF-production.

Pork chain	Novel Protein Food chain
Primary production	Primary production
Crop processing	Crop processing
Feed fabrication	
Animal production	NPF fabrication
Animal processing	
Food fabrication	Food fabrication

Table 2. Environmental impacts of agriculture.

Agricultural activities	Environmental impacts	Impaired regulatory ecosystem components
Land conversion & land use	Habitat loss	Biodiversity
Energy use	Global warming	C-cycle
N-fertilization	Acidification	N-cycle
	Ozone destruction	
P-fertilization	Eutrophication	
Tillage	Erosion	
Irrigation	Water depletion	Hydrological cycle
Pest control	Water pollution	

Based on Helms and Aiking (2003)

Indicators for Monitoring Environmental Relevant Trends of Food Consumption

Jungbluth, N., Frischknecht, R.

**ESU-services, Kanzleistrasse 4, CH-8610 Uster, T. +41 1 940 61 91 or +41 1 940 6132, F. +41 1 940 61 94, jungbluth@esu-services.ch, frischknecht@esu-services.ch*

1. Abstract

Indicators are necessary to compare the state of sustainable development in different countries and to compare the relevance of consumption patterns between different countries. Indicators are also used for monitoring the success of political decisions in the time. This paper develops indicators for monitoring the environmental impacts of food consumption patterns. The research work has been initiated by the OECD. It starts with a review of indicators already proposed by the OECD. Environmental impacts of food consumption can be analysed with different environmental assessment methodologies. Results from the review of different case studies are used to propose a set of indicators that covers all types of environmental impacts related to food consumption patterns in different countries. The indicators are easy to calculate based on statistical data. They cover important environmental aspects like agricultural production methods applied, transportation patterns, consumption levels of product categories, household behaviour and other non-quantifiable key issues.

Keywords: indicator, food consumption, trends, environmental impacts, sustainable development

2. Introduction

Indicators to measure the progress towards sustainable household consumption have been proposed by the Working Group on the State of the Environment (1999). Only general food consumption trends were included in the indicator set (consumption in kg/capita/year and percentage change 1970-1995). In an actual study (Jungbluth & Frischknecht 2000a, b) additional indicators are suggested that could be used to improve the monitoring of environmental impacts from household food choice, preparation and disposal. Furthermore related data sources are identified.

3. Critical review of indicators proposed by OECD

This chapter discusses indicators proposed by OECD for household food consumption. Table 1 shows these indicators (Working Group on the State of the Environment 1999). The two proposals, type of food and growing method, cover some of the important problems.

The type of food is especially important with respect to the level of meat consumption and the comparison of the total consumed food products. The consumption in kg per capita for different product groups can be linked with environmental indicators, e.g. MIPS (Loske & Bleischwitz 1996), EF (Wackernagel *et al.* 2000), energy use and CO_2 (Kramer & Moll 1995), or information from an LCA valued with an impact assessment method (Jungbluth

2000) to compare the environmental impacts over a period of time or for different countries. The examples show the differences in the relevance of different product groups.

Table 1. Indicators for household food consumption proposed by OECD.

Proposed indicators	Comments
Food consumption intensities and patterns: 1. by type of food (fish, meat, etc.) in kg per capita, as % of total 2. by growing method and level of process (share of processed food, share of organically grown produce over total agricultural produce consumed) as % of total	• Reflects a) consumer choices related to food categories and to growing methods, b) shifts in demand towards organically grown agricultural products • To be read in connection with information on the environmental effects of the various growing and production methods (including effects on e.g. fish stocks) • Needs further investigation on policy relevance, and on actual implications for the environment

The indicator for the growing method describes developments in agriculture and food processing that influence the level of environmental impacts caused by fertilizers, pesticides and soil degradation. This indicator needs some clarification on the production methods distinguished (e.g. what is processed food and how should different types of processed food be distinguished). The calculation of environmental impacts would be possible if these indicators would be multiplied with results of one of the methods described. Thus, it is for example possible to estimate the average energy use for different types of food and to combine this with indicator 1. But, it would probably need some research work to estimate the indicator values per kg.

4. Proposal of indicators for monitoring environmentally relevant trends of food consumption

A proposal for household consumption indicators with special focus on food consumption patterns in different countries is now elaborated. These indicators should help to quantify and compare the environmental impacts due to food consumption patterns in different countries over different years. The indicators are discussed in theme blocks for e.g. production methods, transports, etc. Main focus while determining the indicators is laid on (see also Working Group on the State of the Environment 1999:8):

- Policy relevance for monitoring and priority setting.
- Analytical soundness of the indicator with regard to the environmental impacts.
- Measurability and proposal for appropriate methods to quantify the environmental impacts (roughly ordered according to their appropriateness and the data availability).

4.1 Production methods

1. *Share and per capita availability of products from e.g. organic, integrated, conventional and greenhouse production*

This indicator aims to measure shifts in agricultural production practice. It should cover indigenous products as well as imports. The higher the share of products from intensive production like greenhouses the higher is the environmental impact. The indicator serves to control the success of political measures like labeling schemes or environmental subventions for the agricultural sector. Agricultural and foreign trade statistics are the basis for this indicator.

2. *Share and per capita consumption of food products with different degrees of processing (fresh, chilled, conserved, deep-frozen, pre-prepared, ready made, self-service and restaurant)*

This indicator aims to measure the shifts in consumption patterns towards more processed food products. A shift from fresh to conserved and pre-prepared products leads to a rise in energy use and environmental impacts. Statistics for per capita food availability and sales of food retailers and restaurants should serve as a basis for the calculation of this indicator.

3. *Total energy use per capita and the share of different economic sectors (chemical industry, agriculture, food industry, retailers, restaurant, freight carriers, households) for meeting the food demand*

In countries with a good database for input-output statistics this indicator might monitor more directly the impact due to developments in different stages of processing. And it helps to identify the overall importance of different stages in the life cycle of food products.

4. *Percentage of actors in the food chain that have implemented an environmental auditing or management scheme for their company*

An environmental audit helps to identify possibilities for the reduction of environmental impacts. This indicator might also be considered in an indicator set for different industries, as it is not directly related to food consumption patterns.

5. *Food products produced with genetically modified organisms*

The use of genetically modified organisms in agriculture is a theme of critical debate. Today it is difficult to assess the environmental impacts due to an increased use of these organisms and it is unclear how to weight negative or positive aspects.

4.2 Transportation

6. *Per capita average distance and mode of transportation for domestic food transports*

This indicator should measure the domestic transports of food products. A shift from train to road and air based transports and a rise in total transports for food products per capita indicate a rise in environmental impacts. The success of policy measures like energy taxes or reduction of subsidies (e.g. for road transports) can be controlled directly with this indicator. Average transport distances can be investigated with national transport statistics using WASD (Carlsson 1997).

7. *Per capita average transport distance and transport modes of imported food products*

This indicator should measure the impacts due to transports of imported food products. A shift from train to road and air based transports and a rise in total transports for food products per capita indicate a rise in environmental impacts. The indicator helps to identify environmental impacts due to globalization and diversification of consumption patterns.

4.3 Purchasing

8. *Share of different Eco-labels for food products sold in a country*

Eco-labels help consumers to buy less environmentally harmful products. Information about the sales share of different labels helps to measure the acceptance of these labels. Only labels with widely accepted guidelines that show a considerable improvement in comparison to conventional production.

9. *Types of food distribution (direct on farm, market, small shop, supermarket, fast-food, restaurants, etc.)*

The share and frequency of visiting different types of food dealers indicates environmental impacts due to e.g. home transports, land use and construction of buildings. Large supermarkets as well as farms that sell their products directly are often accessible only by private cars. A high share of supermarkets indicates a dissipation of areas for living, shopping and working in big cities. This leads to increased environmental impacts due to transports and land use.

4.4 Consumption level

10. *Per capita food availability (kg or MJ nutrition value per head) and share of different product categories (meat, vegetables, grains, fats, beverages, etc.) in food consumption*

The availability (production + imports - exports) of food differs from country to country. If the availability is higher than the actual demand this leads to food wastes, to over consumption or to long time storage with negative health effects and/or unnecessary environmental impacts. This indicator serves to compare the level of food availability and consumption in different countries. As environmental impacts vary among different product groups this indicator can also analyze environmental impacts due to changing consumption patterns, e.g. a rising share of meat products. Statistical data might be available from foreign trade, agricultural and consumption statistics.

11. *Food availability against food consumption as an indicator for wastage or per capita food waste from waste statistics*

It is possible to assess the share of non-consumed products directly if data for food availability (from agricultural and foreign trade statistics) as well as for food consumption (nutrition studies) are available. The amount of food waste might also be investigated from national waste statistics. The edible parts should be differentiated according to the way of treatment (incineration, deposition, composting, etc.).

4.5 Household behaviour

12. *Per capita packaging wastes, recycling quotas and means of waste treatment for different materials like glass, paper, metals or plastics*

The amount and type of packaging wastes from food products and the way to take care for them is one indicator to assess the environmental impacts due to packaging. A rise in packaging wastes per capita indicates higher environmental impacts while an increased share of recycled wastes indicates an environmental improvement. Data can be gained from national waste and recycling statistics.

13. Mobility for home transport

The share of different transport modes (foot, bike, public transport, car), while buying (food) products, indicates the environmental impact due to home transports. A rising share of private cars indicates rising impacts due to fuel use, noise and land occupation for streets, etc.. Data can be found in transport statistics for the share and distances of different transport modes while purchasing goods.

14. Distribution and energy use of household appliances for food storage and preparation

The number and size of household appliances for food storage and preparation (stoves, freezer, deep-freezer, and small appliances for food preparation) indicates the direct energy use in the household. Data for kitchen equipment can be found in household statistics. The average energy use due to the use of white goods can be assessed.

4.6 Discussion

As the indicators aim to describe the environmental impacts of food consumption over the whole life cycle it is important to reflect not only indigenous food production but to account also for im- and exports.

Some of the indicators proposed are also linked to other fields of household consumption or they might interfere with indicator sets for sustainable development in other economic sectors. The indicator for eco-labelling (0.) might for example also be dealt with in indicator sets for agricultural production practice. Or the share of different modes of transportation (0.) to bring food products to the household might also be covered by an indicator describing the general impacts of private mobility. Prior to a final decision about the indicators it should be crosschecked with proposals for other indicators of sustainable consumption patterns if there are any unnecessary double counting.

5. Conclusions and Outlook

In order to monitor the success of policy measures over a time period and to compare the environmental impacts related to household food consumption in different countries it is necessary to use certain indicators. Different indicators are proposed. In our view out of this the following six indicators are preferred with regard to the data available and the importance of the measured impact:

1. Share and per capita availability of products from e.g. organic, integrated, conventional and greenhouse production
2. Share and per capita consumption of food products with different degrees of processing (fresh, chilled, conserved, deep-frozen, pre-prepared, ready made, self-service and restaurant)

6.+7. Per capita average distance and mode of transportation for domestic and imported food transports

9. Types of food distribution (direct on farm, market, small shop, supermarket, fast-food, restaurants, etc.)

10. Per capita food availability (kg or MJ nutrition value per head) and share of different product categories (meat, vegetables, grains, fats, beverages, etc.) in food consumption
14. Distribution and energy use of household appliances for food storage and preparation

These indicators cover important environmental impacts due to household food consumption and the calculation is based on available statistical databases. The indicators should be cross-checked with indicators from other fields of household consumption or for industries to ensure that there is no double counting of the same effects.

Further research work is necessary after an agreement on the indicators and methods to be used by OECD decision-makers in order to generate generic quantitative factors that can be used to calculate the environmental impacts based on statistical data for different countries.

References

CARLSSON, A. 1997: "Weighted Average Source Points and Distances for Consumption Origin - Tools for Environmental Impact Assessment." In *Ecological Economics* (accepted 1/97).

JUNGBLUTH, N. 2000: *Umweltfolgen des Nahrungsmittelkonsums: Beurteilung von Produktmerkmalen auf Grundlage einer modularen Ökobilanz*. Dissertation Nr. 13499, Eidgenössische Technische Hochschule Zürich, Umweltnatur- und Umweltsozialwissenschaften, dissertation.de, 317 Seiten,www.jungbluth.de.vu, Berlin, D.

JUNGBLUTH, N. & FRISCHKNECHT, R. 2000a: *Environmental Impacts from Household Food Consumption: Methodologies for Evaluation and Indicators for Monitoring.* ESU-services for OECD Environment Directorate, Project on Sustainable Consumption, www.esu-services.ch, Uster.

JUNGBLUTH, N. & FRISCHKNECHT, R. 2000b: *Household Food Consumption Patterns: Part B - Evaluation methodologies and indicators.* Sustainable consumption: Sector case study series No. ENV/EPOC/GEEI(2000)9/ANN1, ESU-services for Organisation for Economic Co-operation and Development (OECD), Environment Directorate, Project on Sustainable Consumption, 49 Seiten, Paris.

KRAMER, K. J. & MOLL, H. C. 1995: *Energie voedt: nadere analyses van het indirecte energieverbruik van voeding.* Final report to the NRP global Air Pollution and Global Change, IVEM research report No. 77, Center for Energy and Environmental Studies of the University of Groningen (IVEM RUG), The Netherlands.

LOSKE, R. & BLEISCHWITZ, R. 1996: *Zukunftsfähiges Deutschland. Ein Beitrag zu einer global nachhaltigen Entwicklung*. BUND UND MISEREOR, Wuppertal Institut für Klima, Umwelt, Energie GmbH, ISBN/ISSN 3-7643-5278-7, 453 Seiten, Birkhäuser Verlag, Basel, Boston, Berlin.

WACKERNAGEL, M., DHOLAKIA, R., DEUMLING, D., RICHARDSON, D. 2000: *Assess your Household's Ecological Footprint 2.0*. www.rprogress.org, San Francisco and Centre for Sustainability Studies, Xalapa.

WORKING GROUP ON THE STATE OF THE ENVIRONMENT 1999: *Towards more sustainable household consumption patterns: Indicators to measure progress.* Environment Directorate, Environment Policy Committee, Organisation for Economic Co-operation and Development (OECD), www.oecd.org/env/consumption/scp26.htm, Paris.

The History Of Bread Production: Using LCA In The Past

Mirja Van Holderbeke[1], *Nieves Sanjuán*[2], *Theo Geerken*[1], *Daniëlle De Vooght*[3]

[1]*Institute for Technological Research (Vito), Integral Environmental Studies, Boeretang 200, B-2400 Mol, Belgium, mirja.vanholderbeke@vito.be*, [2]*Universidad Politécnica de Valencia*, [3]*Free University of Brussels*

Abstract

To get a clear understanding of the process of (un)sustainable development during the past centuries, the environmental, social and economic developments of four basic needs (drinking water, bread, travelling over land and heated accommodation) are investigated both on the production- as well as the consumption side, starting from the pre-industrial period (1800) until the year 2000. For the quantitative environmental assessment, the LCA methodology is used. The project is sponsored by the Belgian Federal Public Planning Service Science Policy. This paper shows the first LCA results and thoughts ensuing from the bread case study.

Keywords: bread, history, LCA, sustainability.

LCA goal and functional unit

The **goal** of the LCA is to compare the environmental effects of producing and distributing bread in the period 1800 till 2000. The **functional unit** is "1 kg representative bread".

Life cycle description of bread for the key years 1800, 1900 and 2000

The specifications given below attempt to represent an average of the way in which the basic need bread was fulfilled for all people at that time.

Description 1800

Rye bread was usually consumed in Belgium. The grain was locally produced and only human or animal energy were used in agriculture. Manure and other organic wastes were applied in low rates to improve the soils. Plant protection was based on cultural measures, e.g. three-course rotation. Transportation to the mill, bakery and consumer was done by push-carts and carts drawn by dogs or horses. Mills operated on natural energies (wind and water). People baked at home in ovens using brushwood. Summarising, only data about the emissions for baking have to be taken into account. Data about the rye yield/ha are extrapolated from figures for other years. The consumption of brushwood and subsequent emissions were determined by emission measurements in a 19th century baker's oven at open air Museum Bokrijk, Belgium. The assimilation of CO_2 in wood and grain was not taken into account. The amount of flour/kg bread was founded on Eiselen (1995).

Description 1900

White bread made of wheat was habitually eaten in Belgium. About 70% of the wheat was imported from the USA by steam trains and steamships, the remaining 30% was produced in Belgium. The use of chemical fertilizers and pesticides became only important after World

War II. Until then, producers depended on the native fertility of the soil to provide the nutrients needed for growth of wheat and on cultural practices and resistant varieties to control weeds, diseases and insects (Paulsen, 2002). Manure was seldom applied (Kansas State Board of Agriculture, 1903) and relating to machinery, no fuel was consumed because all equipment was propelled by farmers or animals. Regarding the energy used in the mills (Bauters, 1998 and own calculations), 41% of the flour was produced by water and wind energy and 59% by steam engines using coal. Transportation of flour to bakeries was done by horse carts. At the beginning of the 20th century, dough was prepared manually and the ovens operated mainly on coal or wood, only a few ovens used gas. The distribution of bread to the consumers was done by carrier-tricycle. Resulting, only data concerning the fuel consumption and subsequent emissions to transport wheat, milling in steam mills and baking have to be taken into account. The yields of wheat crops in Belgium and the USA were founded on NIS (1962) and USDA (2002). The flour yield resulting from and the power needed during the milling process are based on figures from Ammann (1914). The coal consumption in the steam mills was founded on Vierendeel (1921). Emissions (SO_2, CO and CO_2) due to coal combustion were obtained from measurements in Bokrijk. We assume an equal division between coal and wood ovens. The fuel consumption was founded on Ammann (1914). The coal consumption to cross the Atlantic by steamship per kg cargo was calculated using data from SEWSS (2003). Regarding the steam trains, coal consumption was estimated using data from Sinclair (1898). Emissions (SO_2, CO and CO_2) due to coal combustion were based on measurements in Bokrijk.

Description 2000

Nowadays, wheat bread is still the most consumed bread in Belgium. The wheat is mainly locally produced or from European origin. Agricultural processes are mechanised and chemical fertilisers and plant protection products are used to enlarge the yields. Transportation of wheat to Belgian mills is generally done by trains, trucks and ships. Mills operate on electricity and flour is distributed by trucks. Baking processes usually use electricity and heating gas. Transportation of bread to households is done by car, bicycle or by foot. For the LCI, all life cycle steps, from agricultural production until distribution of bread to the consumer, have to be considered.

The wheat used originated in Belgium (30 %), France (43.4%), Germany (19,6%) and The Netherlands (7%) (CLEA—CEA, 2000). The yields of wheat crops in these countries have been taken from FAOstat (FAO, 2003). Data about fuel consumption involved with growing and harvesting of wheat were obtained from Nielsen and Luoma (1999). Sowing rates were taken from Moerschener and Gerowit (1999) and L'Institut du Genech (2002). The amount and type of fertilizers applied to wheat in France, Germany and the Netherlands are based on Ekboir (2002), ITCF (2002) and Moerschner and Gerowitt (2000). Concerning the fertilizers production, data are taken from the SimaPro data base or based on Davis and Haglund (1999). Nutrient balances were founded on L'Institut de Genech (2002), Lopez Bellido (1991), Audsley et al. (1997), Bentrup et al. (2000), Castillon (2003) and Hansen (2000). The use of plant protection products was taken from Eurostat (2002). Regarding the energy requirements for pesticide production, different sources have been consulted (Audsley et al., 1997; Weidema et al., 1995; Ceuterick and Spirinckx, 1997; Van den Broek et al., 2002).

Data about the milling yield, energy and water consumption and prices of flour and by-products were obtained from CERES, the most important mill in Belgium. An allocation based on the weight was made between flour and by-products. Data about water and energy use for baking have been obtained from PRESTI (1996). The amount of flour/kg bread was founded on Andersson (1999). Transportation of wheat to the mill was founded on own assumptions, statistical data and information from CERES. Trucks transport the flour to bakeries (CERES), the average distance taken into account is based on own assumptions. Distribution of the bread to the consumer was established on own assumptions and on statistical data (MOBEL, 2000).

First LCI/LCA results for bread in the years 1800, 1900 and 2000

Figure 1 shows the relative contributions of the life cycle phases in **2000**. The agriculture subsystem is a hot spot for most of the impact categories studied. The potential contributions to global warming are associated with the use of fossil fuels for baking and the fertiliser production for the agricultural phase.

Figure shows the **relative environmental profiles** for the years 1800 and 2000. Surprisingly, the impact categories global warming and photochemical oxidation are much higher for the year 1800, due to the emissions resulting from the brushwood combustion. The baking process nowadays uses four times less energy. Acidification and eutrophication are much higher in 2000, both mainly caused by the production and use of fertilisers.

For the year 1900 not all necessary emission data were available. Therefore we can only compare the emissions of CO, CO_2 and SO_2 for the years 1800, 1900 and 2000 (Figure 3). We can see a large efficiency improvement in the last century. The highest level of these emissions was observed for 1900, followed by the ones for 1800. This is due to the combustion of coal during the transport of grain by steamships and steam trains from the USA in 1900, and to a lesser extent to the combustion processes in the milling and baking phases. At the beginning of the 19th century, all emissions during the bread life cycle were caused by the combustion of brushwood during the baking process. In 2000, CO and CO_2 emissions are attributed to the energy use for fertilizers production, the baking process and distribution to the consumer. However, a significant data uncertainty on this distribution phase exists. SO_2 emissions in 2000 are mainly caused by the energy used in the baking process.

Using LCA in the past: potential, problems and additional assessments

The first LCI and LCA results show already a surprising picture of the environmental impacts accompanying the bread production during the last two centuries. Combining this knowledge with social and economic data will give surely new insights to the sustainability development in the 19th and 20th century. In the economic assessment we will express the number of man hours needed to produce one bread yourself in the past related to the number of man-hours we work today to buy a bread.

However, some practical problems arose during our project sofar. One of the main problems found when applying LCA to the past is the lack and uncertainty of historical data, such as emissions from steam machines or data as manure composition necessary to perform nutrient balances. The emissions from steam machines can probably still be measured.

Another important aspect to take into account when assessing processes in the past are the different kinds of energy sources. In 1800 and 1900 mainly all the energy was of natural, human or animal character. Typically animal and human labor is not included in an LCA, but in this case they are the motor of many stages of the life cycle. The inclusion of an energy indicator (summing fossil, human and animal energy) could be evaluated in order to get an idea of the general energy efficiency improvement.

Another problem is the assessment of land use in the past. We probably will include a simple land use indicator defined as the area of agricultural land needed to produce the amount of grain used to make 1 kg of bread. Its value in1800 is lower (8.1 m^2/ha) than in 1900 (10.5 m^2/ha). This is caused by the more efficient rye milling process in 1800, accounting for a flour yield of nearly 100%. Nevertheless the value obtained for 2000 is the lowest (1.3 m^2/ha), due to the higher yields of the crops and milling process, and also to the protein content of flour that allows to decrease the ratio kg of flour per kg bread.

References

Ammann, L. 1914. Meunerie et boulangerie. Librairie J.-B. Baillière et Fils. Paris France.

Andersson, K. 1999. Life Cycle Assessment (LCA) of Bread produced on different Scales, Case study, AFR-Report 214 CIT Ekologik, Stiftelsen Chalmers Industriteknik, Gothenburg.

Audsley, E. 1997. Harmonisation of environmental life cycle assessment for agriculture. Final report concerted action, AIR3-CT94-2028. Silsoe Research Institute, Silsoe, UK.

Bauters, P. 1998. Van zadelsteen tot zetelkruier. Tweeduizend jaar molens in Vlaanderen, 1. Geschiedenis van het malen met natuurlijke drijfkracht, Gent.

Bentrup, F.; Küsters, J.; Lammel, J.. Kuhlmann, H. 2000. Methods to estimate on-field emissions from crop production as an input to LCA studies in the agricultural sector. International Journal of Life Cycle Assessment, 5(6), 349-357.

Castillon, P. 2003. Personal communication. ARVALIS-Institut du végétal. Baziège, France.

Ceuterick, D. and Spirinckx, C. 1997. Comparative LCA of biodiesel and fossil diesel fuel. VITO report 1997/PPE/R/026.

CLE-CEA. 2000. Landbouwstatistisch jaarboek.

Davis, J. and Haglund, C. 1999. Life Cycle Inventory (LCI) of Fertiliser Production. Fertilisers products in Sweden and Western Europe. SIK-report N° 654. Göteborg, Sweden.

Eiselen Hermann (edt.). 1995. Brotkultur. DuMont Verlag, Köln.

Ekboir, J. (ed.). 2002. CIMMYT 2000-2001 World Wheat Overview and outlook: Developing No till Packages for Small-Scale Farmers. Mexico, DF, CMMYT.

Eurostat. 2002. The use of plant protection products in the European Union. Eurostat, European Comission, Luxemburg.

FAO. 2003. FAOstat, FAO statistical data bases. http://apps.fao.org/cgi-bin/nph-db.pl?subset=agriculture

Hansen, J. 2000. Nitrogen balances in agriculture. http://europa.eu.int/comm/eurostat/datashop/print-catalogue/EN?catalogue=Eurostat&product=KS-NQ-00-016-__-I-EN

Institut de Genech. 2002. http://home.nordnet.fr/~jlnieuviarts/menu/fiches_techniques/productions_vegetales/bletendreraisonne/bletendreraisonne.htm

ITCF. 2002. Blé tendre. Marchés, debouches, techniques culturales, récolte et conservation. Institute Technique des Cereals et des Fourrages. France.

Kansas State Board of Agriculture. 1903. Thirteenth Biennial Report of the Kansas State Board of Agriculture to the Legislature of the State for the years 1901 and 1902. Part IV. Reports of Kansas wheat –growing experience and practice, by counties. Pp. 573-632.

Mobel http://www.mobel.be/

Moerschner, J.; Gerowitt, B. 2000. Direct and indirect energy use in arable farming - an example on winter wheat in Northern Germany. In: Agricultural data for Life Cycle Assessments. Vol 1. Ed: B. P. Weidema and M.J.G. Meeusen. Agricultural Economics Research Institute, The Hague.

NIS. 1962. Nationaal Instituut voor Statistiek , *Landbouwstatistieken, 1900-1961*, Brussel.

Nielsen, V.; Luoma, T. 2000. Energy consumption: overview of data foundation and extract of results. In: Agricultural data for Life Cycle Assessments. Vol 1. Ed: B. P. Weidema and M.J.G. Meeusen. Agricultural Economics Research Institute, The Hague.

Paulsen, G.M. 2002. International contributions to improvement and marketing of Kansas wheat. http://www.kswheat.com/education/facts/history.htm

PRESTI. 1996. Sectoriële studie over de brood- en banketbakkers, ijsberijders en chocoladebewerkers. PRESTI Programma OVAM.

SEWSS, Sucursal Española de World Ship Society, personal communication.

Sinclair, A. 1890. The LOCOMOTIVE ENGINE. Running and Management, Seventeenth Edition -. http://www.railroadextra.com/chapt24.Html

USDA. 2002. Historical track records. United States Department of Agriculture. National Agriculture Statistics Service. http://www.usda.gov/nass/pubs/histdata.htm

van den Broek, R.; Treffers, D.J.; Meeusen, M.; van Wijk, A.; Nieuwlaar, E.; Turkenburg, W. 2002. Green energy or organic food? A life cycle assessment comparing two uses of set-aside land. Journal of Industrial Ecology 5(3), 65-87.

Vierendeel, A. 1921. Esquisse d'une histoire de la technique. Vol 1. Vromant & Co, Imprimeurs-Editeurs. Brussels, Belgium.

Weidema, B.P.; Pederson, R.L. and Drtivsholm, 1995. Life cycle screening of food products. Two examples and some methodological proposals. Danish Academy of Technical Sciences.

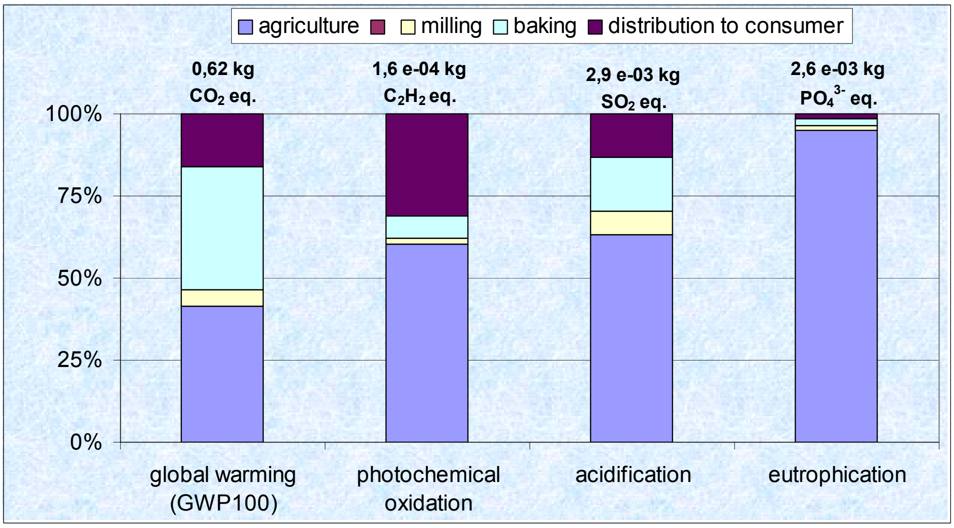

Figure 1. The environmental profile of Belgian bread in the year 2000.

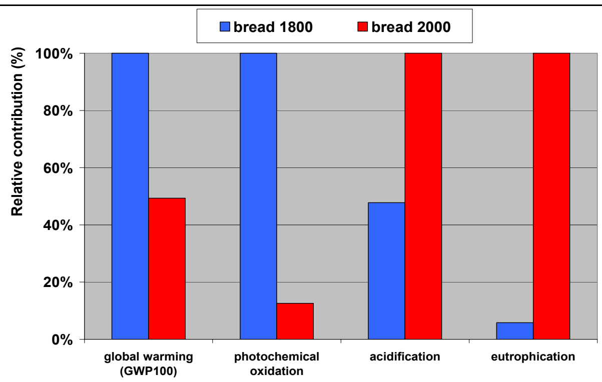

Figure 2. Comparison of environmental profiles of 1 kg bread in years 1800 and 2000.

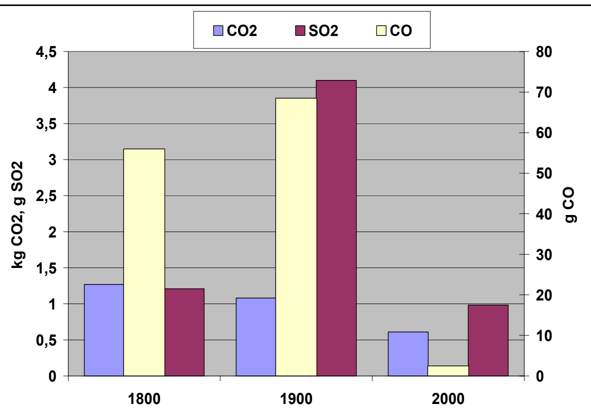

Figure 3. Comparison of CO_2, SO_2 and CO emissions for the years 1800, 1900 and 2000.

Methodological Contributions to tailor Life Cycle Assessment to the Specifics of Arable Crop Production

Brentrup, F.
Hydro Agri, Centre for Plant Nutrition Hanninghof, Hanninghof 35, D-48249 Duelmen, Germany, Tel.: +49-2594-798 137, Fax.: +49-2594-7455, e-mail: frank.brentrup@hydro.com

Abstract
This paper summarizes the main results of a PhD study entitled "Life Cycle Assessment to evaluate the environmental impact of arable crop production" *(Brentrup, 2003a). The study proposes new methodological contributions for 1) the estimation of diffuse, in-field nitrogen emission as an input to the Life Cycle Inventory, 2) the impact assessment of the consumption of abiotic resources like fossil fuels or minerals, 3) the impact assessment of land use 4) and the aggregation of the different environmental impacts into summarizing environmental indices via normalization and weighting. The single modules were integrated into a comprehensive LCA approach, which was then tested in a case study. In this case study the environmental impact of different nitrogen fertilizing intensities in wheat production was analysed.*

Keywords: nitrogen emissions, resource consumption, land use, weighting, wheat production, fertilizer.

Introduction

The Life Cycle Assessment (LCA) methodology provides the theoretical framework for the evaluation of the environmental impact of products or production systems, and thus, can also be applied on arable crop production. This is particularly important since agriculture is expected to comply with the principles of sustainability, which include the three core elements *economy*, *society* and *environment*. In order to evaluate the sustainability of different agricultural production systems, it is necessary to have appropriate indicators for all of these elements in place. Principally, LCA is able to provide those indicators for the environmental aspect.

However, since most of the available ready-to-use LCA models were not specifically designed for agricultural applications they show some difficulties when for instance applied to arable production. Therefore, this study presents contributions to the LCA methodology in order to tailor LCA more to the specifics of arable crop production. These contributions concern the inventory as well as the impact assessment phase and are briefly described in the following (see Fig. 1).

Results

Methods to estimate diffuse, in-field nitrogen emissions as an input to LCA studies
Based on a literature study, structured methods for the estimation of diffuse, in-field nitrogen emissions (ammonia, nitrous oxide, nitrate) were selected. For LCA studies including arable farming it is particularly important to derive reasonable estimates of these highly variable

emissions. The selected methods consider important soil, climate and management parameters (Brentrup et al., 2000).

Conceptual considerations on the impact assessment of abiotic resource consumption
In contrast to other approaches, this study suggests to treat the consumption of resources, which are not substitutable by each other, as separate environmental problems. A final aggregation of non-equivalent resources like phosphate rock and fossil fuels into a summarizing resource depletion indicator is found to be only possible after an explicit weighting procedure, which has been developed in this study (Brentrup et al., 2002a).

Impact assessment of land use based on the 'Hemeroby' concept
Arable farming uses huge quantities of land for crop production. An assessment of the environmental impacts of land use in LCA has to include two dimensions: (a) the size of an area used for a certain period of time and (b) the potential of a specific land use type to degrade the naturalness of the area under use. Whereas the first aspect can be directly expressed as a physical quantity, the latter aspect needs an appropriate indicator. The Hemeroby concept provides such an indicator, since this concept was specifically developed in order to evaluate the level of naturalness of land area. Hemeroby is a measure for the human influence on ecosystems, which defines the level of naturalness of different land use types (e.g. urban area or extensive pasture) according to their deviation from a natural reference situation. This study employs the Hemeroby concept in order to assess the impacts of different land use types within LCA (Brentrup et al., 2002b).

Aggregation into environmental indices
An evaluation of the different environmental effects that are relevant to arable production regarding their potential to harm the environment is performed in order to enable an aggregation of the separate indicators per effect into two summarizing environmental indicators: (a) for abiotic resources and (b) for impacts on ecosystems and human health. This weighting was realized by a comparison of the current status of each effect with defined target values for the respective effects ("distance-to-target principle"). This study suggests internationally agreed environmental targets to be employed in this procedure because they represent a consensus of science, economy and society (Brentrup et al., 2003b).

Case study on wheat production at different fertilizing intensities
After these methodological developments the method was tested in a case study. In this case study the environmental impact of different N fertilizer rates in winter wheat production was analyzed (Brentrup et al., 2003c). The case study revealed that the aggregated environmental impact per ton of wheat grain increases clearly at N rates exceeding the crop demand and at zero N fertilization (see Fig. 2). In the first case aquatic eutrophication was the major problem, whereas in the latter case this is land use. From reduced to economic optimum N rates the environmental indicator values increased rather slightly. At economic optimum N fertilization (192 kg N/ha) aquatic eutrophication contributed most to the aggregated indicator; ter-

restrial eutrophication, acidification, climate change and land use show similar contributions to the aggregated value.

References

Brentrup, F., Küsters, J., Lammel, J., Kuhlmann, H., 2000. Methods to estimate on-field nitrogen emissions from crop production as input to LCA studies in the agricultural sector. *Int. J. LCA* **5**, 349-357.

Brentrup, F., Küsters, J., Lammel, J., Kuhlmann, H., 2002a. Impact Assessment of Abiotic Resource Consumption - Conceptual Considerations. *Int. J. LCA* **7**, 301-307.

Brentrup, F., Küsters, J., Lammel, J., Kuhlmann, H., 2002b. Life Cycle Impact Assessment of Land Use Based on the Hemeroby Concept. *Int. J. LCA* **7**, 339-348.

Brentrup, F., 2003. Life Cycle Assessment to Evaluate the Environmental Impact of Arable Crop Production. PhD thesis, Univ. of Hanover, Germany. 187 pp.

Brentrup, F., Küsters, J., Lammel, J., Kuhlmann, H., 2003b. Environmental impact assessment of agricultural production systems using the life cycle assessment methodology. I. Theoretical concept of a LCA method tailored to crop production. *Europ. J. Agronomy*, in press.

Brentrup, F., Küsters, J., Lammel, J., Kuhlmann, H., 2003c. Environmental impact assessment of agricultural production systems using the life cycle assessment methodology. II. The application to N fertilizer use in winter wheat production systems. *Europ. J. Agronomy*, in press.

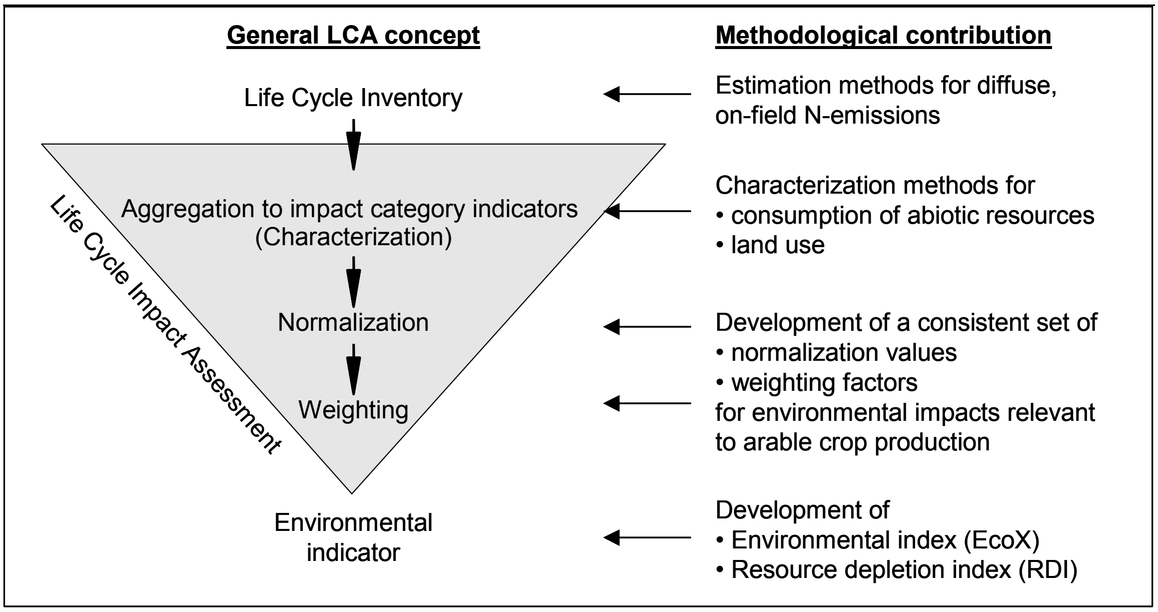

Figure 1. LCA concept and methodological contributions proposed in this study.

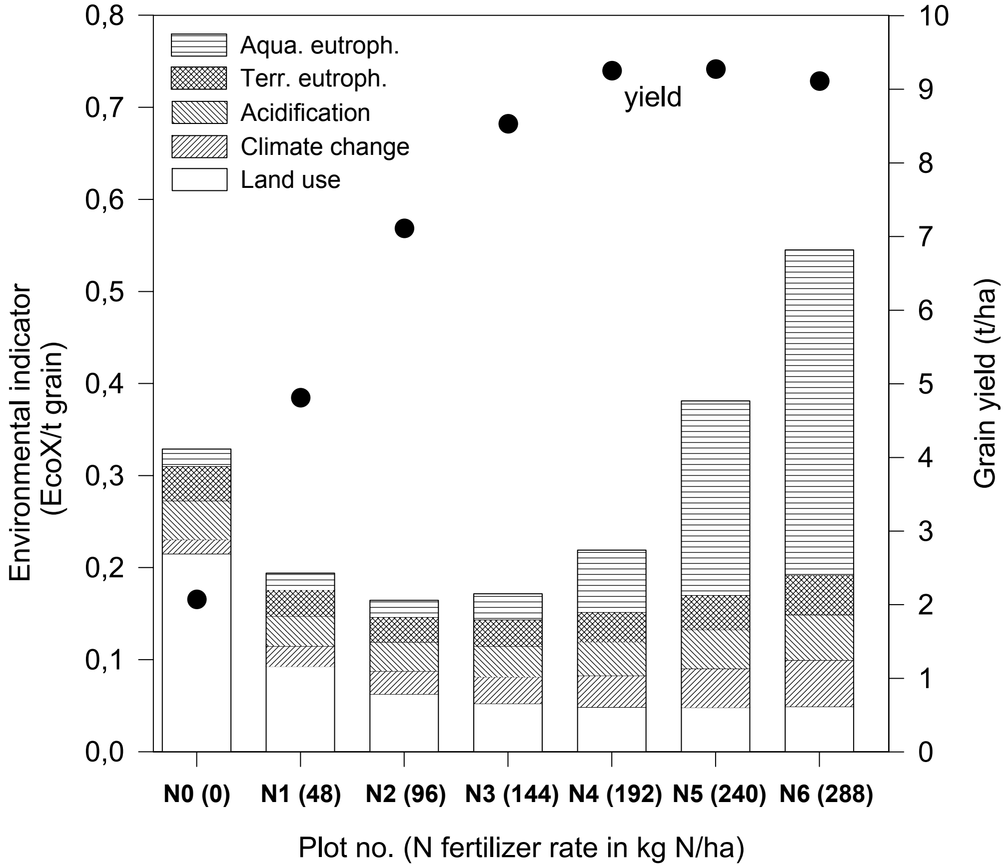

Figure 2. Aggregated environmental indicator values (EcoX) per ton of grain (stacked bars) and yields (t/ha, dots) at increasing N fertilizer rates.

Life cycle inventory modelling in the Swiss national LCI database ecoinvent 2000

Jungbluth, N., Frischknecht, R.

**ESU-services, Kanzleistrasse 4, CH-8610 Uster, T. +41 1 940 61 91 or +41 1 940 6132, F. +41 1 940 61 94, jungbluth@esu-services.ch, frischknecht@ecoinvent.ch Centre for life cycle inventories in the ETH domain, c/o EMPA, CH - 8600 Dübendorf*

1. Abstract

In late 2000 the project ecoinvent 2000 has officially been launched. Several Swiss federal agencies and nine institutes of the ETH domain agreed on a joint effort to harmonise and update life cycle inventory (LCI) data for its use in life cycle assessment (LCA). The goal is a unified and generic set of LCI data of high quality which is valid for Swiss and Western European conditions. The Centre for life cycle inventories in the ETH domain has developed a central database building on past experiences with a large network-based LCI database developed at ETH Zurich. The database comprises in a first version LCI data from the energy, transport, building materials, chemicals, detergents, paper and pulp, waste treatment and agricultural sector (ecoinvent data v1.0). The content of the database will be made publicly available via the web page www.ecoinvent.ch. A data exchange format (ECOSPOLD) has been developed for the data im- and export to different software tools. This format uses the extended mark-up language (XML) for data exchange. The consistent and coherent LCI datasets for basic processes make it easier to perform LCA studies, and increase the credibility and acceptance of the life cycle results.

Keywords: ecoinvent, life cycle inventory, database, background data.

2. Motivation and problem setting

Up to now, several public Life Cycle Assessment (LCA) databases exist in Switzerland, partly covering the same economic sectors (Frischknecht *et al.* 1996, Gaillard *et al.* 1997, Habersatter *et al.* 1998). However, life cycle inventory data for a particular material or process available from the databases often do not coincide and therefore the outcome of an LCA is (also) dependent on the institute working on it. Furthermore the efforts required to maintain and update comprehensive and high quality LCA-databases are beyond the capacity of any individual institute.

At the same time, LCA gets more and more attention by industry and authorities as one important tool for e.g., Integrated Product Policy, Technology Assessment or Design for the Environment. In parallel with this increasing trend in LCA applications the demand for high quality, reliable, transparent and consistent LCA data increased as well. Only a few publicly available LCI databases fulfil these criteria and most of them were published in the nineties.

3. Goal of ecoinvent 2000

That is why LCA-institutes in the ETH domain (Swiss federal Institutes of Technology (ETH) Zürich and Lausanne, Swiss Federal Laboratories for Materials Testing and Research (EMPA) St. Gallen and Dübendorf, Paul Scherrer Institute (PSI) Villigen, and the Swiss Federal Institute for Environmental Science and Technology (EAWAG)) as well as the LCA-department of the Swiss Federal Research Station for Agroecology and agriculture (FAL) in Zurich agreed on a close co-operation. Together with the Swiss Agency for the Environment, Forests and Landscape (SAEFL or BUWAL), the Swiss Agency for Energy (BFE) and other agencies they founded the Centre for Life Cycle Inventories in the ETH domain. The database comprises LCI data from the energy, transport, building materials, chemicals, paper and pulp, waste treatment and agricultural sectors (see Table 1).

A large, network-based database and efficient calculation routines are required for handling, storage, calculation and presentation of data and are developed in the course of the project. These components partly take pattern from preceding work performed at ETH Zurich (Frischknecht & Kolm 1995).

4. Basic structure of the ecoinvent database system

The ecoinvent 2000 database system consists of the following components (see Figure 1):

1. The central database,
2. Calculation routines,
3. The local databases,
4. The administration tool,
5. The query tool,
6. The data (exchange) format,
7. The editor.

Ad 1. The central database contains LCI data and Life Cycle Impact Assessment methods such as the Swiss Ecological Scarcity 1997, Eco-indicator 99 or the CML characterisation scheme 2001. The database is located on a server and accessible via Internet.

Ad 2. Data will be supplied by the partner institutes as non-terminated unit processes. The computation of cumulative inventory results is performed with powerful calculation routines related to the central database. Input data as well as calculations will include (cumulative) uncertainty ranges.

Ad 3. Commercially available LCA-software such as Gabi, SimaPro, Team and Umberto are used as local databases. These local databases are suited for an implementation and use of ecoinvent data. The ecoinvent data (exchange) format is recommended for that purpose.

Ad 4. The administration tool supports the integration of datasets delivered by the co-operating institutes into the central database. It helps to verify the completeness of datasets, calculates inventories and (normalised and weighted) category indicator results and ensures the accessibility for clients respecting the users' rights.

Ad 5. The Query tool is used to interrogate the database and to download datasets from the central database. It enables the search for individual processes, for processes of a certain economic sector (e.g., transport or energy sector) or for data from a certain institute. General information (so-called meta information) about the processes (technology, age, geographic coverage, *et cetera*) is accessible to everybody whereas the quantitative LCI data is only accessible for registered members (clients) of ecoinvent database.

Ad 6. The data exchange format lists all data fields that need to be completed when data is imported into the central database for the first time. It has evolved from the international SPOLD data exchange format (Weidema 1999) and takes pattern from the committee draft of the international technical specification ISO 14048 (International Organization for Standardization (Iso) 2001). Some of the data fields are mandatory, i.e. information must be provided. Among other features, the data exchange format allows for specifying upper and lower estimates (or the coefficient of variance) as well as the probability distribution (e.g., lognormal).

Ad 7. The local administrators use the editor and EXCEL software to create new datasets and to change, enlarge or delete existing datasets. The editor administrates the module names (via a direct link to the central database, where the index of module names is placed). The editor acts as the interface between the local administrator and the central database and generates files in the ecoinvent 2000 data format.

5. Quality guidelines ecoinvent 2000

In the next section selected aspects of the quality guidelines for life cycle inventory analyses performed within the ecoinvent 2000 project are described.

5.1. Introduction

The creation of one central life cycle assessment database requires a high degree of co-ordination and harmonisation. Besides structural aspects and naming conventions, content-related aspects have been discussed and unified. This guarantees a maximum degree of consistency of process data available in the database. Here we focus on content-related aspects.

5.2. Content-related aspects

While structural aspects and naming rules are in many cases controllable by the software, a consistent application of content-related rules is less straightforward. Nevertheless, clear rules are required in order to minimise differences caused by individual, unsystematic choices of the LCI practitioners involved.

System boundaries are drawn based on expert knowledge and not based on fixed rules such as mass or energy shares. If the emission of a pollutant must be expected but no data are available, estimates are used in order to identify whether or not this pollutant may be environmentally relevant.

Electricity is supplied on high, medium and low voltage with increasing losses and investment requirements. Hence electricity demand of processes must be linked to the correct (or most likely) voltage. The supply mix (as well as the export mix) is calculated based on the domestic production plus the imports. In cases where the electricity mix actually purchased deviates from the average supply mix of a nation (or region) such specific mixes (or particular power plant technologies) are used in the model.

Standard transport distances are applied for materials such as steel, cement, basic chemicals *et cetera*, in case the exact distances are unknown. A similar approach is chosen for waste treatment processes. If no particular information is available, standard waste treatment processes defined per material are applied. It is supposed that inert materials go to landfill, plastics are incinerated and metals are recycled.

Allocation is an ubiquitous issue that calls for a harmonised approach. A cut-off approach is used for recycled materials and for by-products (outputs with no economic value that are not sent to waste treatment but are used in other processes). No burdens and no requirements of a preceding process chain and of a process are allocated to the recycled materials and by-products, respectively. On the other hand no benefits are granted for any subsequent use of recycled material or by-product. No fixed prescriptions are made for joint product allocation (co-products) except that system expansion (especially the "avoided burden"-concept) is not recommended.

Fossil and renewable carbon are distinguished for CO_2-, CH_4- and CO-emissions. For renewable energy sources and materials an equal amount of CO_2 is registered as a resource consumption according to the binding capacity of the corresponding crops. Carbon that is emitted as CO is considered when calculating CO_2-emissions. On the other hand, CO will get a global warming potential assuming its subsequent conversion to CO_2.

Uncertainty of flow data is quantified on the level of unit processes. If uncertainty is not known (because not stated in the sources used or because not known by the company providing the data) a standardised procedure is used for estimations. A data quality matrix has been developed which takes pattern from the pedigree matrix published by (Pedersen Weidema & Wesnaes 1996). Scores from 1 to 5 are given for reliability, completeness, temporal correlation, geographical correlation, further technological correlation and sample size. Fixed uncertainty factors are attributed to each of the scores and an additional basic uncertainty is attributed to categories of exchanges (such as electricity and thermal energy consumption, groups of combustion emissions, waste treatment requirements and the like). In most cases a lognormal distribution is assumed. With the help of this standardised uncertainty factors, the geovariance is determined for each individual exchange in the unit processes.

6. Outlook

The software system presented in this paper is in its final development phase. The database is fed with LCI datasets and life cycle impact assessment methods. The size of the economic part of the matrix (up to 3'000 unit processes in its first version) certainly poses a real challenge for the project team in terms of database response and computation time.

The use of XML technology for the exchange of data between the participating institutes and between them and the central database is a challenge not only for the project team but also for the LCA software suppliers who are encouraged to implement the ecoinvent data exchange format. Variations of the data exchange format are possible thanks to the flexibility of the XML technology. This should further enhance the acceptability of this format in the LCA community.

In fall 2003 the database will go online. By then, LCI data with the reference year 2000 will be available via www.ecoinvent.ch for many basic products and services (such as energy supply, transportation and waste treatment services, building materials, chemicals and agricultural products) that make part of most LCI process networks.

References

Frischknecht, R., Bollens, U., Bosshart, S., Ciot, M., Ciseri, L., Doka, G., Dones, R., Gantner, U., Hischier, R., Martin, A. 1996: *Ökoinventare von Energiesystemen: Grundlagen für den ökologischen Vergleich von Energiesystemen und den Einbezug von Energiesystemen in Ökobilanzen für die Schweiz.* Auflage No. 3, Gruppe Energie - Stoffe - Umwelt (ESU), Eidgenössische Technische Hochschule Zürich und Sektion Ganzheitliche Systemanalysen, Paul Scherrer Institut, Villigen, www.energieforschung.ch, Bundesamt für Energie (Hrsg.), Bern, CH.

Frischknecht, R. & Kolm, P. 1995: "Modellansatz und Algorithmus zur Berechnung von Ökobilanzen im Rahmen der Datenbank ecoinvent." In Schmidt M., S. A., *Stoffstromanalysen in Ökobilanzen und Öko-Audits*. Pages**:** 79-95, Springer-Verlag, Berlin Heidelberg.

Gaillard, G., Crettaz, P., Hausheer, J. 1997: *Umweltinventar der landwirtschaftlichen Inputs im Pflanzenbau.* FAT-Schriftenreihe No. 46, Eidg. Forschungsanstalt für Agrarwirtschaft und Landtechnik (FAT), Tänikon.

Habersatter, K., Fecker, I., Dall`Acqua, S., Fawer, M., Fallscheer, F., Förster, R., Maillefer, C., Ménard, M., Reusser, L., Som, C. 1998: *Ökoinventare für Verpackungen.* Schriftenreihe Umwelt No. 250, 2. korrigierte und aktualisierte Auflage, Bundesamt für Umwelt, Wald und Landschaft, Bern, Schweiz.

International Organization for Standardization (ISO) 2001: *Environmental Management - Life Cycle Assessment - Data documentation format.* Technical Specification ISO/DTS 14048 Geneva.

Pedersen Weidema, B. & Wesnaes, M. S. 1996: "Data quality management for life cycle inventories - an example of using data quality indicators." In *Journal of Cleaner Production* Vol. **4** (3-4): 167-174.

Weidema, B. K. 1999: *SPOLD '99 format - an electronic data format for exchange of LCI data (1999.06.24).* SPOLD, www.spold.org.

Table 1. Database content, responsible institutes and their partners in LCI data compilation (see www.ecoinvent.ch for addresses and responsible persons).

Database content	**Responsible Institute**	**Partners**
Energy supply Fuels Heat production Electricity production	Paul Scherrer Institute (PSI)	ESU-services
Plastics Paper and Board Basic Chemicals Detergents Waste treatment services	Swiss Federal Laboratories for Materials Testing and Research (EMPA)	Doka Ökobilanzen, Chudacoff Öko-science
Metals Wood Building materials Basic chemicals	Swiss Federal Laboratories for Materials Testing and Research (EMPA)	Chudacoff Öko-science
Transport services	Swiss Federal Institute of Technology Zurich, (ETHZ UNS)	
Basic chemicals	Swiss Federal Institute of Technology Zurich, (ETHZ LTC)	
Agricultural products and processes	Federal Research Station for Agroecology and Agriculture, (FAL)	Research Station for Agricultural Economics and Engineering (FAT)

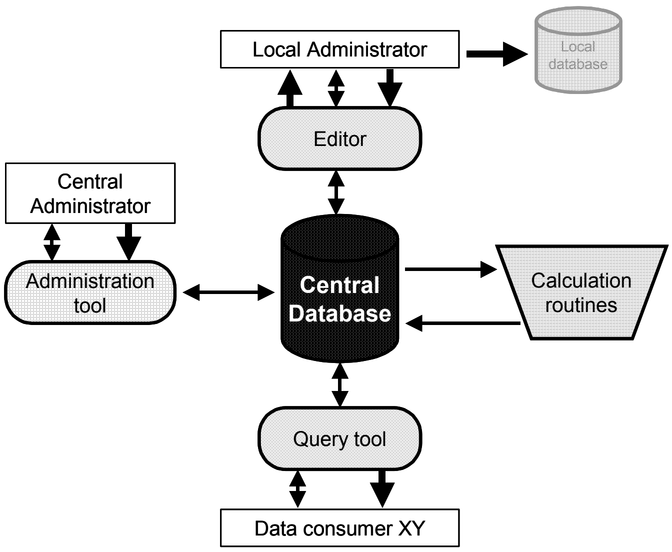

Figure 1. The basic structure of ecoinvent database system.

Environmental profiles of foods for the Swedish market

Lagerberg-Fogelberg, C.[a)], Carlsson-Kanyama, A.[b)], Fogelberg, F.[c)] and Brandén, I.[d)]*

[a)] Swedish University of Agricultural Sciences and Mid Sweden University, Centre for Sustainable Agriculture, P. O. Box 7047, SE-750 07 Uppsala, Sweden, telephone: +46 18 67 16 48, Fax: +46 18 67 35 71, e-mail: Charlotte.Lagerberg@cul.slu.se

[b)] Environmental Strategies Research Group, FOI, SE-172 90 Stockholm, Sweden, e-mail: carlsson@fms.ecology.su.se

[c)] Swedish University of Agricultural Sciences, Dept. of Plant Science,P.O. Box 44, SE-230 53 Alnarp, Sweden, e-mail: Fredrik.Fogelberg@vv.slu.se

[d)] Mid Sweden University, Dept. of Natural and Environmental Sciences, SE-831 25 Östersund, Sweden, e-mail: Ingela.Branden@mh.se

Keywords: agriculture, energy, GHG, wholesale, LCA, processing, resource, retail.

The wholesale and food service sectors control food delivered to large numbers of consumers. About 5.6 million meals are served each day in Swedish institutional households, for example in private and public restaurants in schools, military defence and public health care. The purchasers of food service institutions and the food wholesale sector have a key position in the food chain (Figure 1) in that they control large flows of foods, thereby allowing them to practice potential influence on the food supply chain. Consequently these sectors have a great impact on the overall environmental effects of food consumption patterns.

Environmental profiles of a number of foods are estimated from a life cycle perspective. Supply chains are currently being mapped for the analyzed products. We focus on fresh carrots, tomatoes, apples and meat, frozen broccoli, onion and chicken and dried beans/peas.

Preliminary findings suggest complex patterns of delivery of foods for the wholesale and food service institutions sector. The companies participating in the study buy foods from a wide variety of companies. Companies supplying goods and services are both large and small and several are specialized in trading and logistics. As illustrated by Figure 2, the complexity is not only displayed by the many actors and choices at each level but also by the cross flows of products, i.e. companies in several cases purchase a certain product from more than one source while the upward flows of the product passes through several companies. Consequently the product may eventually be of the same origin but delivered through different supply chains.

The above project is a part of the ongoing three-year project “Designing and evaluating the impacts of environmental information in food service institutions and the food wholesale sector” which is funded by the Foundation for Strategic Environmental Research, MISTRA. The project focuses on how different factors interact in food purchase and how environmental information of foods affect the decisions of food purchase of institutional households and wholesalers. The project investigates how well-designed environmental information can improve environmental performance in food service institutions and the wholesale sector. Needs and practices of suppliers, producers and corporate consumers in relation to environmental information about food are investigated. Simplified descriptions of life-cycle related environmental impacts from selected food products are developed. The focus is on product groups where substantial differences between product choices may be expected. Different types of environmental information and how they interact with factors such as norms and directives will be tested. Recommendations for design of information for specific key users, such as purchasing managers in food service institutions and the wholesale sector will be addressed. The overall potential impact of such information packages on the environmental performance of the studied sector will also be evaluated.

For more and updated information please visit the project homepage at www.e-info.se.

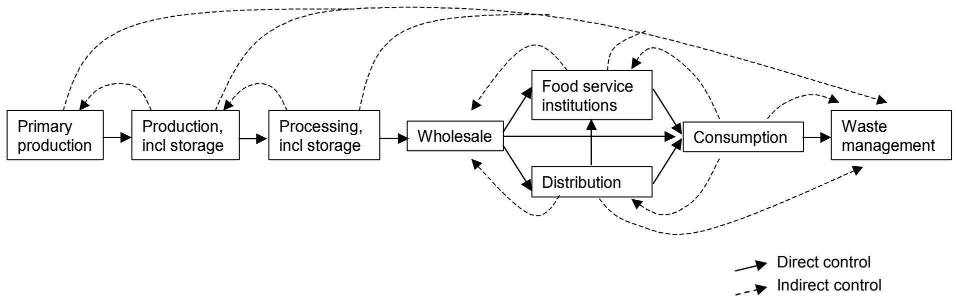

Figure 1. Generic food supply chain. Transportation occur between all compartments.

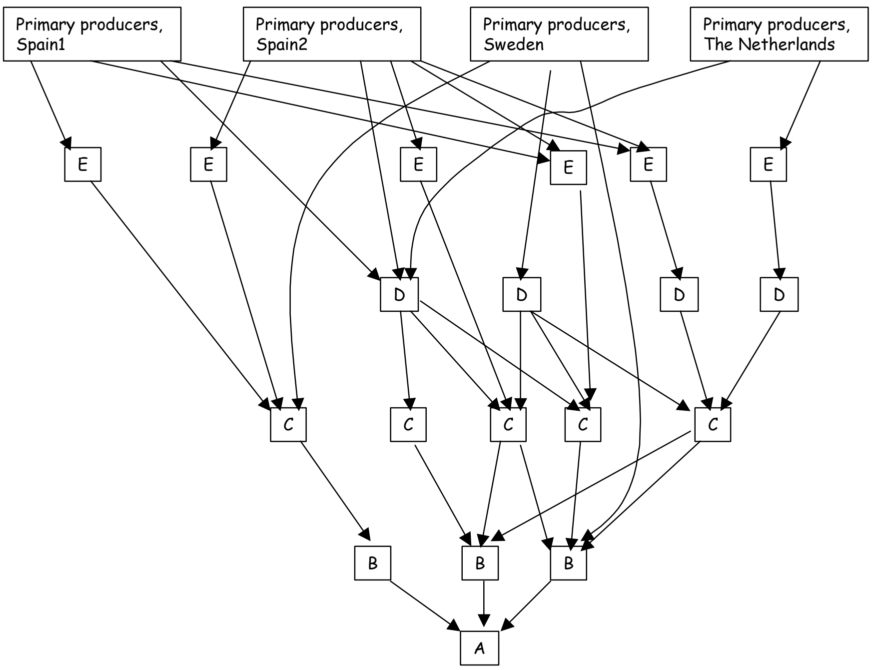

Figure 2. Preliminary supply chain for fresh table tomatoes for one purchasing manager (A).

Using I/O data to find hotspots in LCA - Example of a hamburger meal

Jacob Madsen and Suzanne Effting
PRé Consultants, Plotterweg 12, Amersfoort, The Netherlands, 3821 BB Tel +31 33 4555022, fax. +31 33 4555024 email: madsen@pre.nl, effting@pre.nl

Introduction

Each country has an economic input output table that specifies the value of the purchases between sectors within the country and abroad (the imports), as well as the supplies to other sectors and the exports. Furthermore all other major costs and revenues are specified. These tables have been used by several LCA experts to compile input output (IO) databases [1].

In the present study we compare the environmental impact of a hamburger meal based on data from traditional LCA process databases and data from the Dutch Input Output database containing sector IO data both from the Netherlands and the rest of the world. Since LCA is an iterative process it is essential to have data available of your full life cycle. When performing an LCA you first identify the hotspots in the life cycle and the results you use in the refinement of your study. The advantage of IO data is that it covers an entire economy thus including all products. However IO data lack process specificity since all products are aggregated into sectors. With this in mind the objective of this study is to validate the data from the IO database in order to fill out the gaps that normally occur in the data collection process.

Databases

In our work we aim to use existing databases to provide a preliminary picture of the environmental impact of the product under study. A simplified model of the life cycle of a hamburger meal was build using process data based on Buwal [2] for the packaging and IVAM [3] for the food products that go into the hamburger meal i.e. Cola, French fries and a hamburger sandwich. At the same time an identical model was built using I/O data from the Dutch I/O database [4].

The Dutch I/O database

The Dutch input output database consists of a matrix with 105 producing sectors and their imports. Regarding emissions the Dutch "emission registry" system maintains a detailed data inventory for industrial activities. This database has been used to run queries that produce datasheets per sector. In order to calculate the impact of imported products, the "rest of the world" was split up into three regions:

- OECD countries in Europe
- Other OECD countries
- Non OECD countries

For each of these regions, thirty sectors were defined that were taken from the DIMITRI [5] and EDGAR [6] database. The EDGAR database already has data on Energy use, CO2, NOx and SOx per country and per sector. To cover the other stressors a wide range of sources has been consulted. In order to focus the efforts, an analysis was made using the GTAP [7] database to identify which countries or regions contribute most to an industrial activity. The focus was to find data for these countries and regions first, and extrapolate this data over the whole region. Of course the data collection was not complete, and often extrapolations have had an important influence.

The use of different sector definitions for the Netherlands (105 sectors) and the three regions that cover the rest of the world (30 sectors each) requires a conversion routine.

An aggregation table has been constructed that specifies which of the 105 Dutch sectors can be aggregated to one of the 30 international sectors.

Each Dutch sector has 105 domestic purchases and 105 imports. The 105 imports were converted into 30 imports using the aggregation table.

Dutch trade statistics have been used to determine which share of each import comes from the 3 regions which are used to model the world outside the Netherlands. Competing and non competing imports have been treated in the same way. Also here a 30 sector aggregation has been used.

Inventory
An average hamburger meal was estimated to consist of:

- 83 gram beef
- 88 gram bread
- 40 gram tomatoes, salad etc
- 0,7 kWh Electricity
- 225 gram potatoes
- 376 gram cola
- 20 gram paper
- 20 gram plastic
- 100 km truck transport
- 7,5 MJ heat

The life cycle was modelled in SimaPro 5.1 and calculated with the Eco-indicator 99 method.

Results
When comparing the results of the process LCA with the IO LCA we observe the same pattern. The categories land use, fossil fuel use and respiratory inorganic substances dominate the overall environmental impacts, see figure 1.

However when looking at the process contribution we noticed that the I/O data for potatoes, tomatoes and salad have a significant environmental impact (figure 2). The agricultural sector is very heterogeneous and the average product from this sector is probably not representative for products like potatoes, tomatoes and salad. Additionally, the average product from the 'animal based food sector' underestimates the impact from cow meat when comparing the two data sources since cow meat typically has a higher impact in comparison with for example chicken and pork.

Furthermore we notice a significant environmental impact from the services related to the production of a hamburger meal. In this context services are marketing, advertisement etc. The impacts from services are something that seldom is taken into account in traditional LCA.

Conclusions

The results showed that the I/O database provides representative data for identification of the hotspots in the life cycle of a hamburger meal. However, when examining the process contribution we discovered that the environmental impacts from potatoes, tomatoes and salad were high. Additionally the environmental load from cow meat was low compared to process LCA data. The agricultural sector is a heterogeneous sector and the average product from this sector is not representing products like tomatoes, salad etc. Therefore, additional data collection regarding these inputs would be needed. This also supports the general assumption that IO LCA is complete in system boundaries but lack process and product specificity. At the same time, LCA's based on process databases often are specific and detailed but incomplete due to cut-offs. One solution is the hybrid method where you combine the two methods. We recommend to use available process LCA data and I/O to fill in the data gaps and complete the inventory of the whole life cycle.

References

1. Joshi, S.; Product Environmental Life-Cycle Assessment Using Input-output Techniques, Journal of Industrial Ecology, Volume 3, nr 2&3, 2000
2. BUWAL 250, Life Cycle Inventories for Packagings, Volume II. Environmental Series No. 250, Bern, 1998
3. IVAM LCA Data 4. Ewijk, H.A.L., Krutwagen, B.T.J.M., Saft, R.J., Otto, A. IVAM 2002
4. Goedkoop M., Database Manual Dutch Input Output Database 95, PRé Consultants, Amersfoort, 2003
5. Wilting, H.C. et. Al. Dimitri 1.0 : Beschrijving en toepassing van een dynamisch input-output model (*Dimitri 1.0 : Description and application of a Dynamic input-output model, in Dutch)*, RIVM report 778001 005, Bilthoven, the Netherlands, 2001
6. Olivier, J.G.J et al. Description of EDGAR 2.0. RIVM report 771060002, Bilthoven the Netherlands 1996; a recent version 3.0, can be accessed on www.rivm.nl/ieweb/ieweb/index.html?databases/edgar.html.
7. GTAP database, Center for Global Trade analysis, Purdue University, www.gtap.agecon.purdue.edu/

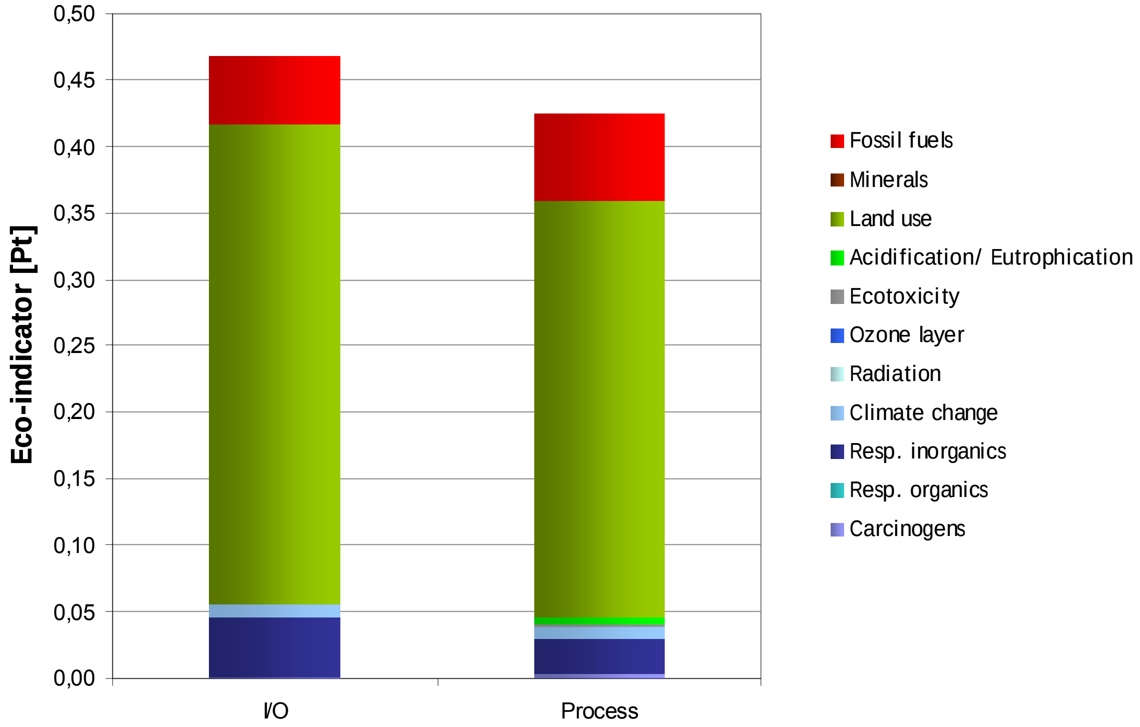

Fig. 1. Comparison of the environmental impact from the life cycle of a hamburger meal when using process LCI data and I/O LCI data.

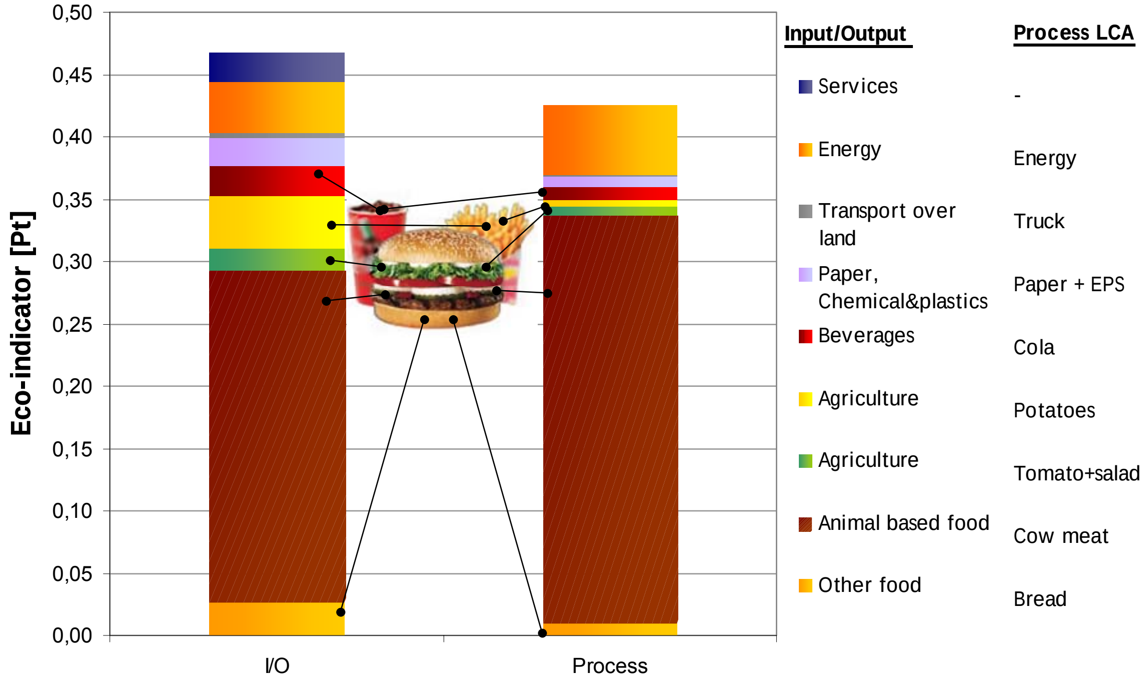

Fig. 2. The process contributions to the environmental impact of the life cycle of a hamburger meal when using process LCI data and I/O LCI data

The use of by-products from food industry as basis for livestock feed and the consequences for the analysis of the environmental impacts of meat consumption

*Nonhebel, S., Elferink, E.V.**
** Center for Energy and Environmental Studies, Nijenborgh 4, 9747AG Groningen, The Netherlands, Tel: (++)31-50-3637177, fax (++)31-50-3637168*

Abstract
The production of meat puts a large claim on resources. The allocation used, however, is of influence on the environmental impact of meat. These allocations can change suddenly due to economic or food safety regulations. Meat production is, however, closely linked with the production of other food products. Focussing on pork only instead of the entire food system makes the environmental impact of pork opaque. Therefore, determination of the environmental impact of the combined production of pork and by-products gives far more consistent results than an analysis of individual products.

Keywords: allocation, meat, environmental impact, by-products, system analysis.

Introduction

Analysis of the environmental impact of food packages has shown that especially the consumption of meat goes together with large environmental impacts. A more detailed analysis of the meat production system, however, shows that a large fraction, over 50%, of the livestock concentrated feed is generated from by-products from the food industry e.g. sugar pulp and molasses from the sugar industry, oil-seed cakes from the vegetable oil industry, but also slaughter wastes (Dutch statistical bureau of agriculture and horticulture, 2000). This large fraction of by-products in the feed complicates the analysis of the environmental impacts of meat. When these by-products are considered as unwanted 'waste-streams' of the food industry the environmental impacts allocated to meat are very low, and the production of meat can be considered as an environmental friendly method to convert a by-product in a highly valued food item (Nonhebel, 2003). In that case meat production even prevents dumping wastes.

Presently by-products from the food industry can be considered as co-products, since the food industry obtains a considerable income on selling their 'waste-streams' to the livestock fodder industry. However, new insights in food safety and economical trends have led and will lead to changes in permitted by-products in livestock fodder. BSE in cows, for instance, led to a ban on the use of slaughter waste as an ingredient for livestock fodder. This implies that presently available by-products can change in value from highly valued basis of livestock fodder, to very unwanted by-products. Such changed regulations may have large consequences for the derived environmental impacts of meat.

This paper focuses on the environmental impacts allocated to the by-products, various allocation procedures are recognised and the consequences for the results are evaluated.

Method

System description

Production of food products of animal origin as meat milk and eggs, requires a complex system with interaction at different levels of scale and between systems, e.g. local vs. global and livestock vs. crop. Analysis at a higher level of scale than currently done in environmental assessments, leads to improved insight in the system, the environmental impacts and its major players (Nonhebel, 2003).

Dutch pork production and consumption is studied in this paper as a model sector for western meat production. In the Netherlands pigs are mainly kept in an intensive manner, meaning that they are kept in barns and are fed concentrated feed. Dutch pig production can roughly be divided in three sectors, a pedigree sector, a multiply sector and a fattener sector. After fattening pigs are being slaughtered, processed and consumed. Transportation occurs between all sectors. The fodder industry produces the feed, produced from two streams of raw materials. The first are imported raw materials primarily grown for concentrated feed, e.g. grains, pulses and tapioca. The second are different kinds of by-products from the Dutch food industry. The largest streams of by-products, representing 66% of total available, are; soybean cake, by-products from the sugar industry, beet pulp and molasses, and potato peels and residues (Dutch statistical bureau of agriculture and horticulture, 2000). The by-products are relatively cheap, homogeneous and have a good nutritional value. Therefore these streams are very suitable ingredients in concentrated feed but also for feeding directly to livestock. The nutritional value of fodder for pigs is expressed in Energy value pig (Evp). This nutritional value is determined from literature for each type of raw material (Centraal Veevoederbureau, 1997). Taking loses due to premature dead and cutting into account, 4 Evp is needed to produce 1 kilogram of pork (Elferink, 2001).

The indicator used in this paper is energy input. The energy input is allocated to the by-products until fodder production. It is assumed that energy input in the remainder of the pork production and consumption chain is independent of the raw materials used for fodder, and is therefore left out in this analysis.

Allocations used

There are several options for allocating environmental effects (Proce, 1986). In this study by-products are allocated in three different manners:

1. On a by-product basis, meaning that no environmental impact was attributed to the by-product but only the main product.
2. On the present economical value as a raw material for fodder.
3. On mass ratio

Table 1 shows the allocation ratio for the different crops and concentrate ingredients as used in this study as well as the energy required for yielding the crop and the Evp. The table is derived from data from; the Dutch statistical bureau of agriculture and horticulture (CBS), Agricultural Economics Research Institute (LEI) and the Dutch Central Bureau for Livestock Feeding (CVB).

Results

Table 2 shows the energy input to produce the raw materials for 1 kilogram vegetable oil, sugar and potato product. The energy input for fodder is not on a kilogram base but depends on the amount of by-product released from the food industries to produce 1 kilogram of main product. Results are compared with feeding on wheat only. This is comparable with a scenario in which food regulations forbid the use of by-products in fodder. The amount of grain is equal to the total Evp, 4.8 Evp, of the by-products generated.

If allocated on by-product, fodder has no impact, while the 'main' products have a high impact. This scenario is not realistic because the food industry gains a lot on selling by-products. When allocated on mass the energy input of by-products is slightly lower than for wheat. An allocation on economical value, however, shows an energy impact that is substantial lower for fodder on the different by-products than for fodder on wheat.

The allocation used has consequences for the calculated environmental impact of a product. However, total energy use of al products together, e.g. meat, sugar, etc., is constant as long as the food system doesn't change. When the system changes, however, energy input changes also. In this case total energy input increases when by-products are not being fed anymore.

Discussion and conclusion

The allocation used has a large influence on the energy input required to produce fodder for pork. These allocations can change suddenly due to new (food) regulations or economic trends. Making the environmental impact of pork very opaque. It is further shown that there is a strong correlation between the food production industry and the livestock fodder industry. These industries affect each other in the choice of agriculture products used as basis for their production process. The determination of the energy input of the combined production of pork and by-products will give far more consistent results than an analysis of individual products. This recycling of by-products as analysed in here for pork also occurs in other parts of the food production system. More attention to these product linkages would improve the insight in the environmental impacts of food production system as a whole.

References

Centraal Veevoederbureau, 1997. Voedernormen landbouwhuisdieren en voederwaarde veevoeders; Verkorte tabel. Lelystad.

Dutch statistical bureau of agriculture and horticulture, 2000. Land en Tuinbouwcijfers. 's-Gravenhage, LEI-DLO/ CBS.

Elferink, E. V., 2001. Vlees een duurzame eiwitbron? Een modelstudie naar het inzetten van organische reststromen in de veehouderij. Groningen, RuG, Center for Energy and Environmental Studies (IVEM).

Nonhebel, S., 2003 On resource use in food production systems: The value of livestock as 'rest-stream upgrading system'. *Ecological Economics*. Accepted.

Proce, C., 1986. Energieverbruik in de nederlandse akkerbouw en veehouderij. IVEM-report nr. 17. Groningen, IVEM.

Table 1. Allocation ratios, yields and inputs for the crops and their ingredients used for concentrated pig feed.

Crop	Products	Yield ton/ha)	Energy input (Mj/ kg)	Evp	Allocation (%) mass	price	by-product
Wheat	Grain	8.2	3.10	1.01	100	100	100
Soybean	oil, cakes	2.5	2.95	(1.39) 0.96	20, 80	50, 50	100, 0
Sugar beet	sugar, beet pulp, Molasses	57.9	0.55	(0.26) 0.98 0.80	59, 24, 17	91, 5, 4	100, 0, 0
Potato	Potato, peels	47	0.94	(0.98) 1.21	77, 23	99, 1	100, 0

Table 2. Energy input of agricultural products allocated in four different ways.

Allocation	Oil (MJ/kg)	Sugar (MJ/kg)	Potato (MJ/kg)	Fodder (MJ)	Total (MJ)
by-product	26.3	9.1	3.5	0.0	38.9
price	13.2	8.2	3.5	14.0	38.9
mass	5.3	5.3	2.7	25.6	38.9
grain	26.3	9.1	3.5	19.7	58.6

Is manure separation a solution to environmental problems from large scale pig production?

Claus Grøn Sørensen[1], Niels Halberg[2], Randi Dalgaard[2]

[1]Danish Institute of Agricultural Sciences, Department of Agricultural Engineering, Research Centre Bygholm, DK-8700 Horsens, Tel: 7629 6023. E-mail: Claus.Soerensen@agrsci.dk

[2]Danish Institute of Agricultural Sciences, Department of Agroecology, Research Centre Foulum, P.O. Box 50, DK-8830 Tjele.

There is a rapid structural development of the agricultural holdings in Denmark. It is estimated that the number of dairy farms and the number of pig farms will be reduced by 50% within the next 10 years. However, milk production will be maintained and pork production increased at the same time. This development will result in an increased specialization and a marked enlargement of each holding with respect to land use, substance turnover, and economic turnover. Hereby, the economic and environmental impact of the actions taken by each farmer is increased considerably.

In relation to the structural development, new technologies are introduced and new production systems emerge. These include other ways of handling slurry including biogas production and separation, and other ways of carrying out tillage of the arable land. These technologies influences resource use (energy etc.), nutrient efficiency in the system, use of toxic elements, and eventually environmental impact, (greenhouse effect, acidification, eutrophication, and ecotoxicity) of the agricultural production. Often an introduced technique can have a positive influence in one aspect but a negative influence in another aspect and it may be difficult to make an easy assessment of it.

In order to comply with national priorities and international obligations, there is a need to know how the new production methods may affect the range of environmental impact in question and to identify where in the production process the critical events occur. Also, it is important to make such information available for decision makers at different levels including politicians, farmer organizations, the advisory service, and companies involved in development of the new technologies. This is the objective of the project: Evaluation of innovative agricultural production systems through a life cycle assessment (LCA) methodology (2003-2005).

Overall, the aim of the project is to investigate and evaluate the impact of introducing new production technologies within, pork production, and plant production in resource use, and the environmental impact using an LCA methodology. Moreover, it is the aim to identify the most 'critical' production process or production phases in relation to specified resource use and environmental impact parameters (greenhouse effect, acidification, eutrophication and eco-toxicity) resulting from the production of a specific amount of food-products.

Specifically, it is the aim, within the above context, to evaluate:

1. Different ways and methods of handling liquid manure on pig farms including establishment of on-farm biogas systems and handling of separation fractions.
2. The impact of introducing new precision farming technologies (e.g. new tillage techniques as e.g. no-ploughing) in arable farming on energy use, nutrient efficiency, and pesticide use.

New innovative systems for slurry handling on pig farms, and minimal tillage techniques are identified and formulated in a collaborative process including the advisory service, researchers, and farmer organizations. Each system is 'optimized' in land use as well as in machinery and equipment. The resource use, the technical efficiencies, the emissions, and the changes in carbon soil sink in relation to the agricultural production process will then be measured in private farms or estimated. The systems are evaluated through a life cycle assessment methodology as well as through traditional economic tools. By using the LCA methodology the environmental impact is evaluated per unit of production in agreement with the idea of the efforts in the EU to improve the environmental policy through a product-oriented approach. In addition, estimates of the environmental impact on a per-hectare basis and of farm economy are obtained.

Scenario identification

A number of models of large scale pig production were derived from a participatory workshop with pig producers and representatives from companies involved in manure separation and biogas techniques. Figure 1 shows an example of a future farm with a total production of 83850 pigs on 3 units including sow/piglet production. Danish regulation limits the application of manure to 140 kg total N in manure applied per ha. Therefore, there is an increasing interest in methods for reducing the amount of slurry to be handled and transported. Separation of manure or slurry into liquid and fibre phases may reduce the amount of water transported when exporting nutrients from the pig farm to agricultural land on other farm. However, it is not clear how this effect will reduce the total environmental impact from intensive pig production. Different systems using manure separation and biogas will be studied in private farms and modelled within an LCA framework in order to compare the environmental impacts. The need for additional knowledge as regards methods and operational data for the selected model farms will be determined through a detailed identification of the existing data on systems for separation, biogas, etc. This evaluation will determine the amount of additional data to be collected from private farms and separation installations.

Transport and logistics

As part of the modelling framework, specific models on, for example, the logistics of different manure distribution strategies will be used to evaluate the transport efforts in terms of labour input and transport distances. For example, the average transport distance is estimated for various different scenarios involving amounts of manure, harmonization demands, and separation.

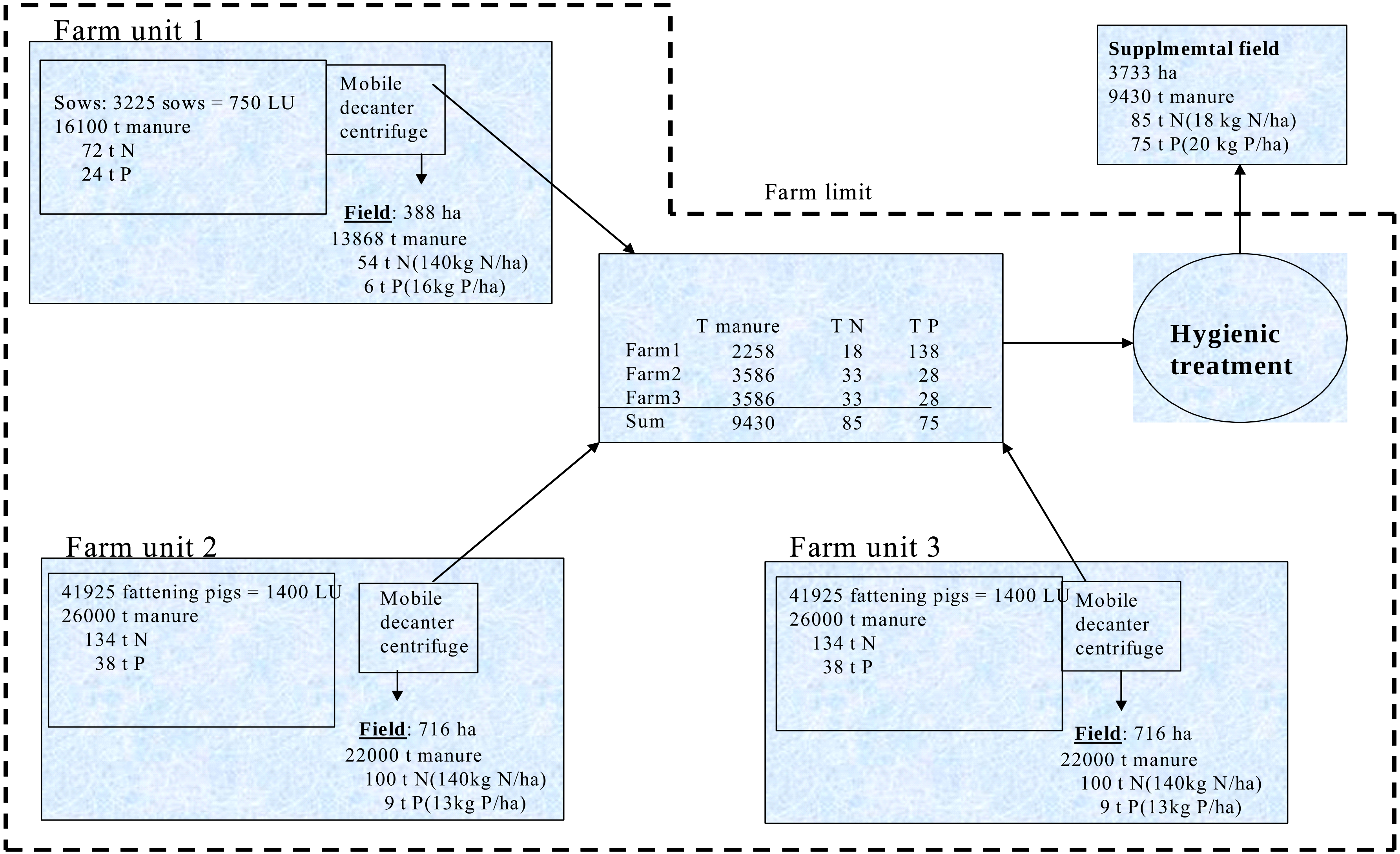

Figure 1. Scenario: low-technology slurry separation.

Separation of slurry entails a number of advantages, depending on the type of separation technology, including lower costs of storage, transport, and application of slurry, as well as better utilization of the nutrients in the field. A typical transport scenario involves: (1) that the liquid fraction from the decanting separation is applied following the N-norm, (2) that a certain amount of the separated solid fraction is applied following the P-norm within the harmonization area supplemental to the liquid fraction, and (3) that the surplus solid fraction is transported out from the farm to a secondary recipient location for application. Figure 2 shows the transport distances for decanting fractions as a function of the distance to “manure-free” areas receptive to manure application and as a function of varying harmonization demands in terms of LU ha^{-1}.

For smaller distances to secondary localities, the average transport distance is only dependent to a small degree on the harmonization demands. The dependency grows when this distance is increased. The transport distance/work is reduced by 61–78% as compared with the baseline scenario with no separation. The modeling suite has the advantage of coping with numerous different farm scenarios and application scenarios. In addition, the results from using the modeling suite may be further elaborated and serve as input for the estimation of economic indicators.

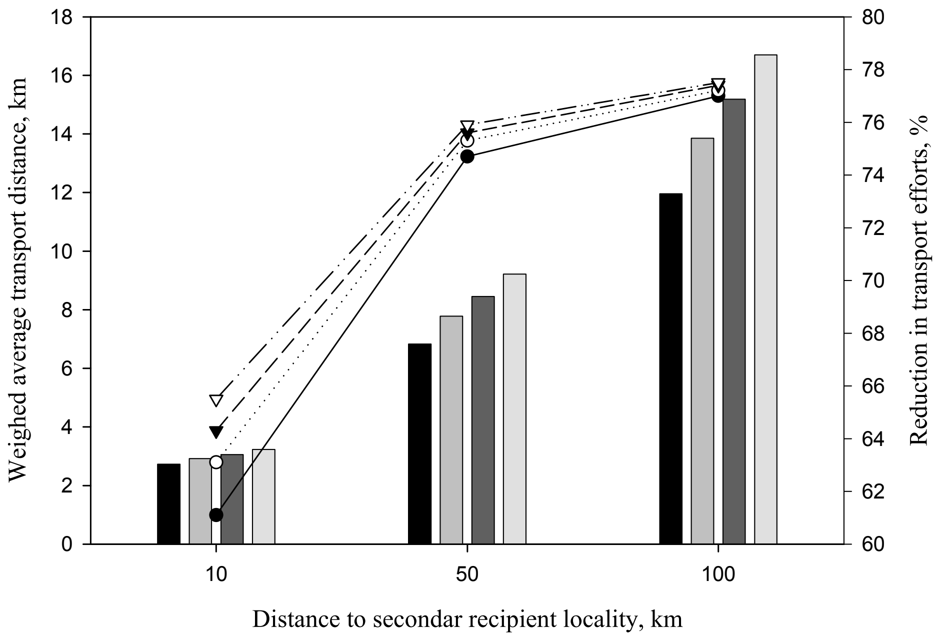

Figure 2. Distance for total transport of manure for application on primary farm land, transport to secondary recipient land, and subsequently application. Default values: 1000 LU, equaling 20 000 t of slurry per year. The dosage for the liquid fraction and the solid fraction from the separation is 180 kg N per ha (utilized 145 kg N per ha) equaling 37 t ha^{-1} and 21 kg P ha^{-1} equaling 3 t ha^{-1}. The reduction of the transport distance is related to a baseline scenario involving no separation and application of raw slurry with a dosage of 21 kg P ha^{-1} equaling 14 t ha^{-1}. The shape of the adjoining land is assumed to be a quarter circle with an area corresponding to the respective harmonization area: ▬, 1.4 LU/ha; ▭, 1.7 LU/ha; ▬, 2.0 LU/ha; ▭, 2.5 LU/ha; ●, 1.4 LU/ha; ○, 1.7 LU/ha; ▼, 2.0 LU/ha; ∇, 2.5 LU/ha.

LCA of Danish milk -system expansion in practice

Randi Dalgaard and Niels Halberg
Danish Institute of Agricultural Sciences, Department of Agroecology, Research Centre Foulum, P.O. Box 50, DK-8830 Tjele. Tel: +45 8999 1254 - Fax: +45 8999 1200. E-mail: Randi.Dalgaard@agrsci.dk

Background

Danish mixed dairy farms do not only produce milk but also several co-products (e.g. meat, bread wheat, manure, rape seed). When estimating the emissions related to milk production it is difficult to decide to what extent the emissions are related to milk and to what extent they are related to the co-products. For example the methane emission from the cows enteric fermentation is both related to the milk and beef production. In most LCA-studies on milk, co-products allocation has been used as a solution to the problem. Co-products allocation implies partitioning of the emissions between the products and the partitioning factor has mostly been based on economic values or mass. Co-products allocation is not recommended as best practice by International Standard Organization (ISO 14041,1998), and can be avoided by use of system expansion, as shown in this study.

Objectives

To make LCA of milk produced on a Danish farm and accounting for co-products by the use of system expansion

Methods

A typology of 31 farm models representing the Danish agricultural sector was established. Inputs (e.g. electricity, soybean meal, fertiliser) to each farm type and emissions were estimated from account data and modelling as described by Dalgaard et al. (2004). Co-products were accounted for by use of system expansion. Milk was considered as the main product on farm types producing milk and the co-products (e.g. beef, sugar beet, bread wheat) were assumed to substitute products on other farms, and thereby contributing with a negative amount of emissions. Data describing resource use and emissions per functional unit of fertiliser, soybean meal etc. was derived from Nielsen et al. (2003).

Results

The processes affected by the production of 1 kg of Danish milk from farm gate are shown in figure 1. These processes can be divided into three main categories: 1) Processes before the farm (e.g. soybean cultivation, grain feed production) 2) Processes on farm (e.g. fodder production, manure handling) 3) Processes after the farm (e.g. avoided production of meat, sugar beet, fertiliser).

The dairy farm type shown in figure 1 has an agricultural area of 48 hectares with grass-clover in crop rotation on 19% of the land. The farm type produces 538 tons milk per year and the average milk yield per cow is 7100 kg milk. It has a high stocking density (2.7 LSU per hectare.)

Production of soybean meal imported to farm
When producing soybean meal there is a co-production of soy oil. Soy oil substitutes rapeseed oil on the world market and it can be assumed less rapeseed will be produced.

Thus a demand for soybean meal increases soybean production and decreases rapeseed production (Weidema, 1999). Soybeans are capable of N-fixation and rapeseeds are not, and therefore increasing soybean meal production lead to a decrease in fertiliser use.

Production of milk on farm
In the system the dairy farm is the major contributor to global warming (1061 g CO_2 equivalents per kg milk). Methane from cows, nitrous oxides from crop residues and manure handling are the most important emissions of greenhouse gasses.

Avoided production of meat
For each kg of milk 44 g of cattle meat is produced. The more expensive parts (tenderloin etc.) replace beef and the cheaper parts replace pork. The avoided production of meat contributes with – 533 g of CO_2 eqv. per kg of milk.

Avoided production of fertiliser
According to Danish legislation a dairy farm with a stocking rate higher than 2.3 LSU per hectare is obliged to sell part of the manure. A part of the Nitrogen and Phosphorus in the manure will replace artificial fertiliser used on cash crop farms. This will again reduce greenhouse gas (GHG) emissions from fertiliser production, which is then deducted from the GHG of the Dairy farm. But the saved GHG emissions from the avoided production of fertiliser is very small, and only contribute with -2 g CO_2 per kg milk.

The farm data used above represents one of 8 farm types divided by soil groups and stocking density to be found in the LCA-food database (Nielsen et al., 2003). Figure 2 shows GHG emissions from four conventional dairy farm types and the part of emissions related to the use of concentrate feed in the form of soybean meal.

Discussion and conclusion
Manure sold from a farm will replace artificial fertiliser as shown in figure 1. Thereby a farm that sells manure gives rise to a decrease in GHG emission from fertiliser production, because it is assumed that the manure-receiving farm will import less fertiliser. But the ammonia loss from manure is much higher than ammonia loss from fertiliser, which means that the manure-producing farm contributes to an increase in ammonia emission on the manure-receiving farm. This emission is therefore included in the environmental impact from milk production in our study due to the use of systems expansion rather than allocation. Two of the dairy types (farm type 6 and 18) in this study sell manure and emissions connected to sold manure is counterbalanced in the LCA database (Nielsen et

sen et al., 2003) used for the analysis. However the counterbalancing has very little impact at the results.

The study has showed that system expansion in agricultural production systems is possible in practice. LCA of milk also showed that the dairy farm is the major contributer to global warming. The results of this method for different livestock products in a number of impact categories may be found on the open LCA database (Nielsen et al., 2003).

A more detailed analysis of the contributions of on-farm processes and use of external inputs is needed to explain the differences between the farming systems. Moreover, the present data structure does not allow to estimate the uncertainty or variation on the emission estimates. Sensitivity analysis should be carried out in order to quantify how important the differences between the farm types are compared to differences in farm practise within the types.

References

Dalgaard, R., Halberg, N., Kristensen, I.S. and Larsen, I. 2004. An LC inventory based on representative and coherent farm types. In: Halberg , N. (ed.) Life Cycle Assessment in the Agri-food sector. Proceedings of 4th International Conference. DIAS report. Danish Institute of Agricultural Science. Denmark (this volume).

ISO 14041. 1998. Environmental management -Life cycle assessment –Goal and scope definition and inventory analysis. International standard. Reference number: ISO 14041: 1998(E).

Nielsen, P.H., Nielsen, A.M., Weidema , B.P., Dalgaard, R. and Halberg N., 2003. LCA food database. Available online, 31/8 2003 at www.lcafood.dk .

Weidema, B. 1999. System expansions to handle co-products of renewable materials. Pp. 45-48 in Presentation Summaries of the 7th LCA Case Studies Symposium. Brussels: SETAC-Europe.

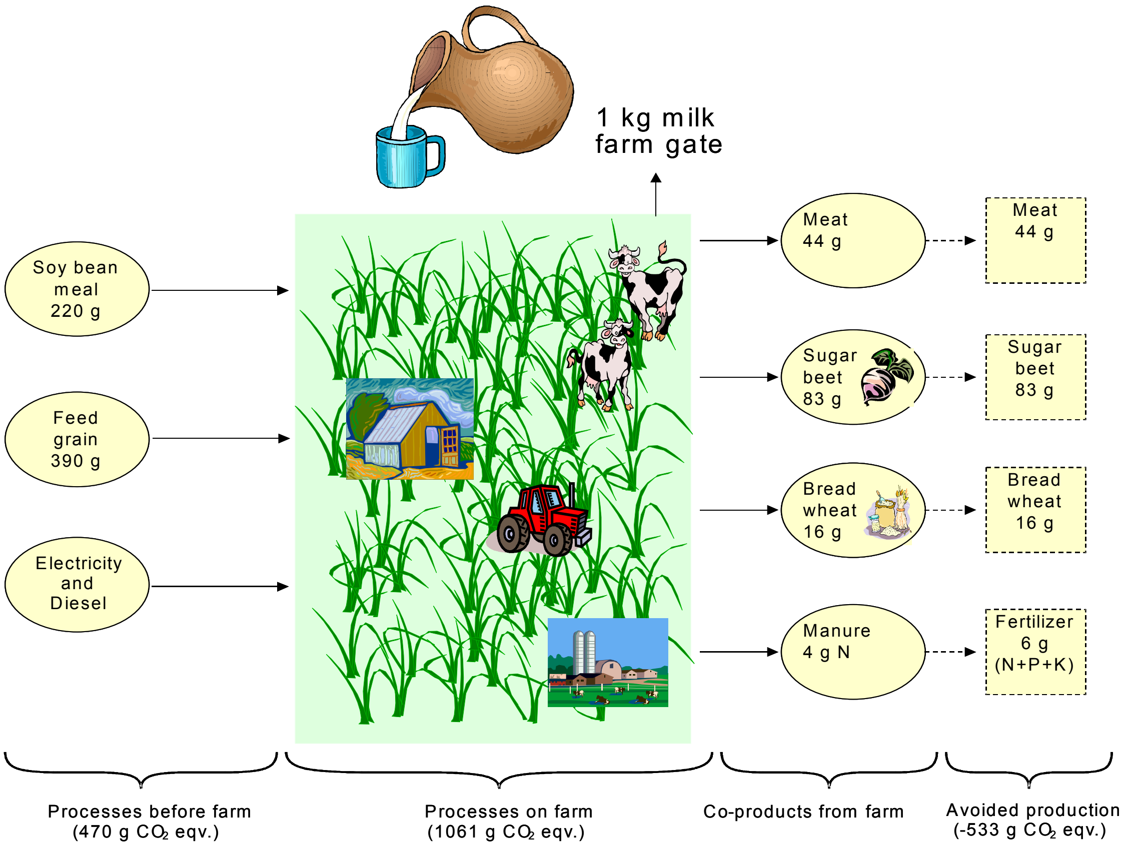

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Figure 1. Processes affected when producing 1 kg of Danish milk on sandy soil. Data derived from the LCA-food database (farm type 18).

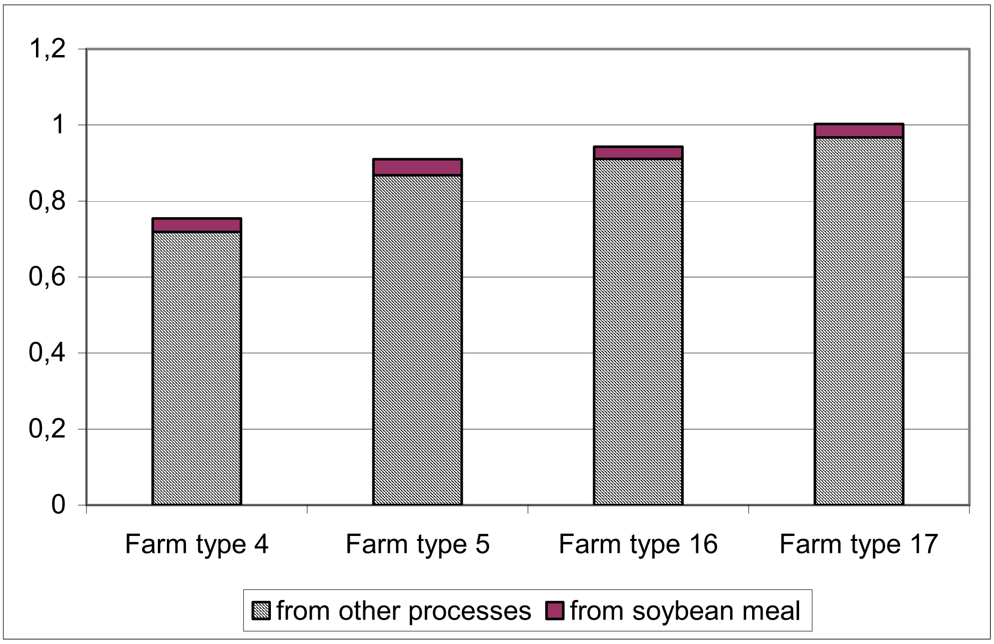

Figure 2. GHG emissions (kg CO_2 eqv.) per kg milk produced at six different farm types. Farm type 4: Loamy soil, 0.9 LSU per hectare. Farm type 5: Loamy soil, 1.7 LSU per hectare. Farm type 16: Sandy soil, 1.1 LSU per hectare. Farm type 17: Sandy soil, 1.8 LSU per hectare.

LCC and LCA of extra-virgin olive oil: organic vs. conventional

Notarnicola B.[*], *Tassielli G.* [*], *Nicoletti G.M.*[**]
**Dipartimento di Scienze Geografiche e Merceologiche, Facoltà di Economia, Università di Bari, via C. Rosalba, 53, 70124 Bari, Italy.*
Tel: 0039 080 5049084; fax: 0039 080 5049019; e-mail: b.notarnicola@dgm.uniba.it
[**]*Facoltà di Economia, Università di Foggia, via IV Novembre 1, 71100, Foggia, Italy.*

Abstract
Olive oil represents a relevant productive sector in Puglia, a region of the South of Italy, since it covers more than the 50% of the whole Italian output and about the 18% of the EU output. In the last years, the production of organic extra virgin olive oil is highly increasing due to a new consumer behaviour and to the high organoleptic, nutritional and healthiness quality of this product. But organic extra-virgin olive oil still remains a niche product because of its market price remarkably higher than the other oil and fats. In this paper the production systems of the conventional and organic extra-virgin olive oil have been compared, in order to assess their environmental and cost profiles, and to verify if the two dimensions – environmental and economical – converge in the same direction.

Keywords: LCC, LCA, extra-virgin olive oil, organic agriculture

1 Introduction

In the last ten years the Italian agricultural area cultivated by following organic practices is remarkably increased, going from a 0.5% of the total area in the 1993 to the 8% in the 2001. The total Italian "organic" area is about 1,200,000 ha, about the 15% more than the previous year, featuring about 60,000 farms. The "organic" area is shared among fodder (56%), wheat (18%), arboreal cultivations (19%) and vegetables (6%) (Compagnoni et al., 2001). Among the area cultivated by arboreals (about 228,000 ha) olive tree is particularly important, since it covers 44,175 ha. The olive oil production of the Southern Italian region Apulia covers about the 50% of the total Italian production and about the 18% of the EU production. In the light of these data and of the importance of the Apulian oliviculture and olive oil production (1182 olive oil factories on the Italian total of 5514), a high rate of growth for the organic oliviculture can be forecasted in this region. Apart the producer countries, olive oil price remains remarkably higher than other oils and fats even if it is characterised by a clear better environmental performance due to the lack of chemical treatments (Nicoletti et al., 2001a). This situation, which is exactly the contrary of what an environmentally friendly policy should do – to support on the market place environmentally friendly products –, becomes much more marked in the case of the organic oil in which, together with the typical olive oil high price, basically due to the cost of labour in the extremely delicate operation of olives harvesting, one has to add the additional costs due to the minor yields (about 30%) of the organic soil. Therefore, even more than the conventional one, organic extra-virgin olive oil is destined to be a niche product. But we think that a superior product from the health, nutritional and environmental point of view should be affordable by everyone. In this study the results of a Life Cycle Costing (LCC) and the first results of a more comprehensive comparative Life Cycle Assessment (LCA) of organic and conventional extra-virgin olive oil are shown in order to identify the relative economical and environmental scores and to verify if the dimensions converge or not in the same direction.

2 Methods

The methodologies which have been used are: the LCC as suggested by the guidelines stated by White (White et al., 1996) which divide the costs into three categories, *conventional company costs* (typical costs which appear in the company accounts), *less tangible, hidden and indirect costs* (less measurable and quantifiable, often obscured by placement in an overheads account) and *external costs* (the costs which are not paid by the polluter, but by the polluted); the LCA as stated by the ISO 14040 rules (ISO, 1996). The functional unit is 1 kg extra-virgin olive oil and the analysis is from cradle-to-gate. The physical and economical data have been taken directly from farms, olive oil factories and by databases. The internal and external costs are respectively shown in Table 1 and 2. The external costs relative to the energy have been taken from the ExternE National Implementation Italian Report (ExternE, 1997), while those relative to the use of pesticides and fertilisers from a study of the Bocconi, Milan, Italy in which the production and social costs of the organic and conventional agriculture have been compared. The study has taken into account the impact of the agricultural activities on the water and has monetarised these impact showing that the damage caused by the conventional agriculture due to fertilisers and pesticides in terms of reclamation and decontamination costs is 33 times superior to that of the organic agriculture.

3 Results and conclusions

As other studies on the organic systems (Nicoletti et al., 2001 b), the results of the LCA (Fig. 1) show that the organic system scores worse than the conventional one in all the impact categories with the exception of the NP, HTP, TETP and FAETP (the impact assessment method used is the CML 2000 as stated in Guinèe et al., 2002, with the exception of the ADP category which is substituted by the EC, Energy Content, and with the addition of LU, Land Use). The minor yield (about the 30%) of the organic system is the reason of this result. By going through the evaluation step (Fig. 2) it can be found out that the organic system results more eco-compatible than the conventional of about 5 times due to the relevant difference in the TETP and FAETP impact categories. Figure 2 shows the differences in the results if one accounts for the external costs. If one does not consider the external costs, the organic oil has a superior cost profile which is basically due to its minor agricultural yields (about the 30% less). On the contrary, by adding the external costs, which are not actually paid by the farmer and by the olive oil companies, to the conventional company and to the less tangible, hidden and indirect company costs, it can be found out that the organic oil has a minor total cost compared to the conventional oil. This result enlightens the need to account for external costs as the European Commission has started to do (Labouze E. et al., 2003). The options for environmental improvement in the conventional system are mainly related to a more reasonable use of pesticides while, in the case of the organic, a reuse of the brushwood as fuel rather than their uncontrolled burning on the fields, could lead to a better environmental profile both in the HTP and in the POCP. Moreover, in the organic system the "traditional" extraction method has been used in the inventory set-up; on the contrary, the AIAB guidelines (AIAB, 2001) permits to the organic oil producer to use the "continuous-extraction method" which is characterised by an energy consumption double than the traditional process. It would be desirable a major attention of these guidelines to the energy consumption, since the consumer who is interested in organic foods would like to buy a more eco-compatible product, which is characterised not only by the absence of the use of fertilisers and pesticides but also by the least energy consumption. On the cost side, Fig. 2 has shown the importance in taking into account the external costs. Since a minor cost of the organic olive oil compared to the conventional one is not obtainable on the market place just with the "market laws", it is necessary to promote tools of public intervention which could end up in reducing the gap between the cost of the conventional oil calculated by the traditional cost accounting methods and those calculated by following the LCC approach. The aim should be that on the basis of the same quality standards, products with a better environmental profile should have a minor market price compared to the concurrent; exactly the contrary of the present situation in which the most eco-compatible products have higher market prices.

References

AIAB (Organic Agriculture Italian Association), 2001. Guidelines for the production of extra-virgin olive oil, (in Italian). www.aiab.it

Compagnoni A., Pinton R., Zanoli R., 2001. Organic farming in Italy 2001. www.organic-europe.net

ExternE National Implementation Italy. Final Report. October 1997. http://externe.jrc.es/it.pdf

Guinèe J.B., Gorèe M., Heijungs R., Huppes G., Kleijn R., de Koning A., van Oers L., Wegener Sleeswijk A., Suh S., Udo de Haes H.A., de Bruijn H., van Duin R., Huijbregts M.A.J., 2002. Handbook on Life Cycle Assessment, Operational Guide to ISO Standards, Dordrecht, Kluwer Academic Publishers.

International Organization for Standardization ISO/DIS 14040, 1996. Environmental management - Life cycle assessment - Principles and framework, ISO/TC207/SC5.

Labouze E., Monier Veronique, Le guern Y., 2003. Study on external environmental effects related to the life cycle of products and services. European Commission, Directorate General Environment.

Nicoletti G.M., Notarnicola B., Tassielli G., 2001a. *Comparative LCA of virgin olive oil vs. seed oil*, Proceedings of the International Conference LCA in Foods, 26-27 April, 2001, Gothenburg, Sweden.

Nicoletti G.M., Notarnicola B., Tassielli G., 2001b. *Comparison of conventional and organic wine*, Proceedings of the International Conference LCA in Foods, 26-27 April, 2001, Gothenburg, Sweden.

White A.L., Savage D., Shapiro K., 1996. Life Cycle Costing: concept and applications. In Curran M.A. (ed.). Environmental Life Cycle Assessment, Mc Graw Hill.

Table 1. Internal costs of the two systems for functional unit (€).

Agricultural phase	**Organic**	**Conventional**
Pesticides	0.171	0.117
Fertilisers	0.268	0.181
Lube oil	0.023	0.011
Electric energy	0.143	0.085
Water	0.077	0.046
Diesel	0.084	0.048
Labour	4.344	2.864
Organic certification costs	0.064	-
Total	**5.174**	**3.352**
Transports	**0.078**	**0.039**
Industrial phase		
Electric energy	0.014	0.024
Labour	0.089	0.045
Water	0.002	0.022
Packaging	0.298	0.298
Waste authority	0.015	0.015
Organic certification costs	0.009	-
HACCP certification costs	0.0009	0.0009
Total	**0.428**	**0.405**
Total	**5.680**	**3.796**

Table 2. External costs of the two systems for functional unit (€).

	Organic	**Conventional**
External costs of energy	0.664	0.533
External costs of fertilisers and pesticides	0.439	9.870

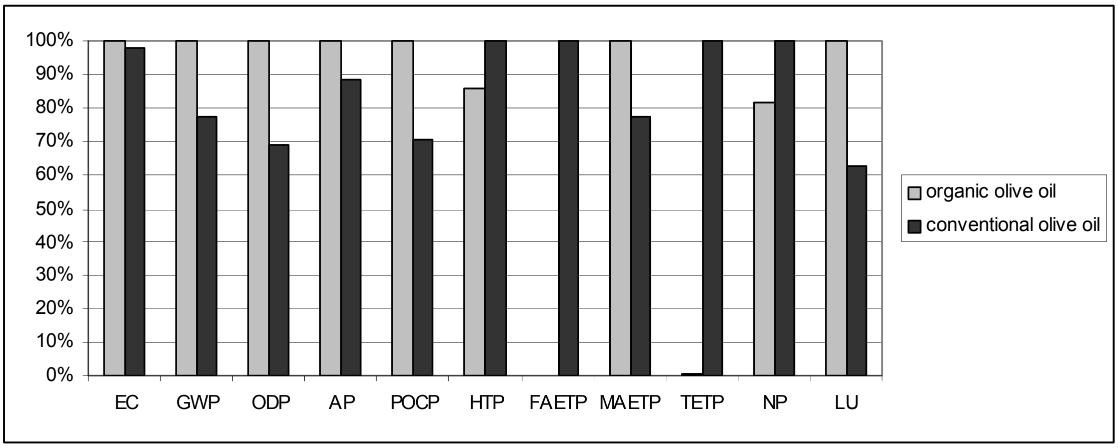

Figure 1. LCA results.

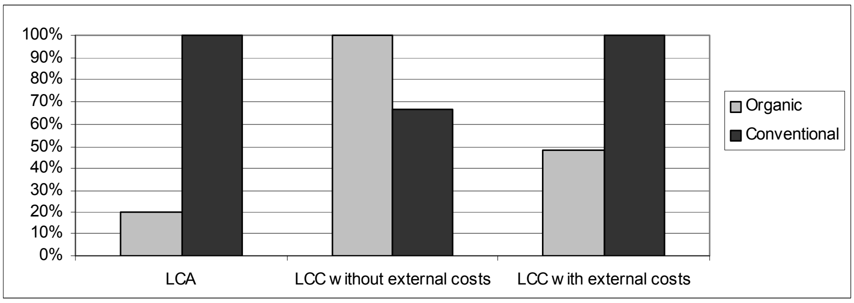

Figure 2. LCA-LCC without external costs and with external costs.