


Book of Proceedings

the 5th International Conference

LCA in Foods

25 – 26 April 2007, Gothenburg, Sweden

Organised by SIK – The Swedish Institute for Food and Biotechnology
www.sik.se/lcafoods2007

Life cycle assessment of imported agricultural products – impacts due to deforestation and burning of residues

Jungbluth, N., and Frischknecht, R.

ESU-services Ltd., Kanzleistrasse 4, CH-8610 Uster, T. +41 44 940 61 91 or +41 44 940 6132, F. +41 44 5445794, jungbluth@esu-services.ch

Swiss Centre for life cycle inventories, Empa, CH - 8600 Duebendorf, frischknecht@ecoinvent.ch

Abstract

Life cycle assessment (LCA) of imported oil plants like soybeans or plant oils, e.g. palm oil, is an important issue in many studies for food products and animal production. Quite often, imported products are assessed with the same data as national products. Country specific aspects for the location of production might thus be forgotten.

In an ecoinvent project for the investigation of biofuels several agricultural products have been investigated. The aim of this project was to investigate data for biomass production, conversion to biofuels and use for transport services. The production of fuels like ethanol, rape seed methyl ether, BTL (biomass-to-liquid), etc. is investigated in a way consistent with the existing ecoinvent datasets.

The findings from this project are quite interesting also for studies on food products. The presentation highlights the relevant methodological issues for global biofuel production, like accounting for CO_2 emissions due to land transformation and clear cutting of tropical rain forests. Results from the LCA study for soybeans and oil produced in Brazil and the US, sugar produced in Brazil as well as for palm oil production in Malaysia are presented. The assessment shows that CO_2 and particle emissions due to deforestation and burning of harvesting residues might form an important part of environmental impacts throughout the life cycle. Especially the issue of deforestation should be taken into account for countries with increasing agricultural production area.

Keywords: imported food products, ecoinvent, life cycle inventory, deforestation

1. INTRODUCTION

The extent of environmental impacts of food products depends on various factors, e.g., agricultural technique, transport distance or packaging.

In an ecoinvent project, biofuels from several agricultural products have been investigated [1]. The aim of this project is to investigate data for biomass production, conversion to biofuels and use for transport services. The production of fuels like ethanol, rape seed methyl ether, BTL (biomass-to-liquid), etc. is investigated in a way consistent with the existing ecoinvent datasets [2]. Different types of fuels are compared in a second part of the project [3].

Life cycle assessment (LCA) of imported oil plants like soybeans or plant oils, e.g. palm oil, is also an important issue in many LCA studies for food products and animal production. Quite often imported products are assessed with the same data as national products. Country specific aspects for the location of production might thus be forgotten.

The presentation focuses on the methodology for integrating such aspects and on some examples.

2. METHODOLOGY

Several aspects of modelling have to be considered for the unsustainable use or deforestation of primary tropical forests and its following transformation to agricultural or forestry land. Due to the initial felling, carbon dioxide is released from burning and degradation of unused biomass. Later on, bounded carbon dioxide in the wood is released after its use. Thus it should be considered as a CO_2-release. A second source of CO_2-emissions is the release of carbon bound in the soil. The bounded carbon is degraded after the transformation i.e. to agricultural land. All CO_2-emissions due to land transformation from wood burning and degradation of carbon bound in soil are recorded with a new type of emissions.

The emissions must be allocated among first initial felling with the production of wood and the following use as agricultural or forestry land. Therefore a multi-output dataset is inventoried. First, the land is transformed to “forest, clear-cutting”. If no better information is available, all carbon dioxide releases from burning of wood and degradation of soil-bounded carbon content are allocated to the use of the land for agriculture or forestry [1].

Forest and grassland conversion is the major cause for CO_2-emissions in Brazil. A total of 951 billion tonnes of CO_2 have been emitted in 1994. This equals 92.4% of the total CO_2-emissions in the country. Also other pollutants like carbon monoxide, methane, N_2O, etc. are released in important shares due to the land conversion activities. In Malaysia about 7.6 Mio. tonnes of CO_2 are released due to land conversion activities. This equals about 7.8% of the national CO_2 emissions in 1994.

In Malaysia about 150000 ha/a are provided for palm fruits. In Brazil about 2 Mio. ha/a are provided for soybeans. The total increase of area for one production period is considered for the production that takes place during the year 2005.

Among 151 to 190 tonnes of carbon per hectare are bound in the biomass above the ground. The degradation of this bounded carbon depends on the subsequent use of the area. All details are elaborated and described in an ecoinvent report [1].

3. RESULTS

3.1. Soybean production

Figure 1 shows the results for the indicator “greenhouse gas emissions”. Without taking the emissions from clear cutting into account soy beans produced in Brazil would have about the same global warming potential as those produced in the USA and a much lower figure than these produced in Switzerland. The picture changes considerable, if the emissions from land transformation are taken into account. The emissions from soybeans produced in Brazil are more than doubled. A similar result can be found for soybean oil from Brazil and palm oil produced in Malaysia.

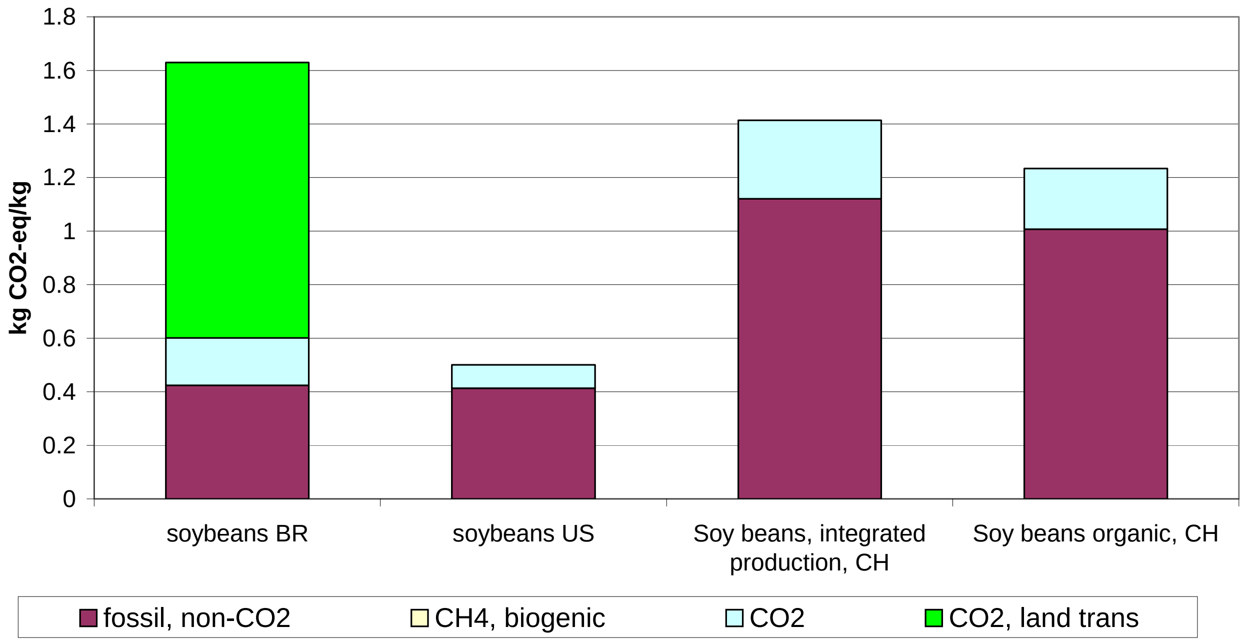


Figure 1 Emission of greenhouse gases for the production of 1 kg soybeans (kg CO2-eq/kg)

The effect of clear cutting the primary forest prior to soybean production is not only important for the release of greenhouse gases, but also for other environmental impacts. Wood residues are burned after clear cutting. During this combustion particles, CO and NMVOC are emitted. Figure 2 shows a life cycle impact assessment with the new Swiss ecological scarcity method [4]. In the case of soybeans produced in Brazil, particle and benzene emissions are quite important. The evaluation also shows further differences in the production patterns e.g. due to the use of different pesticides.

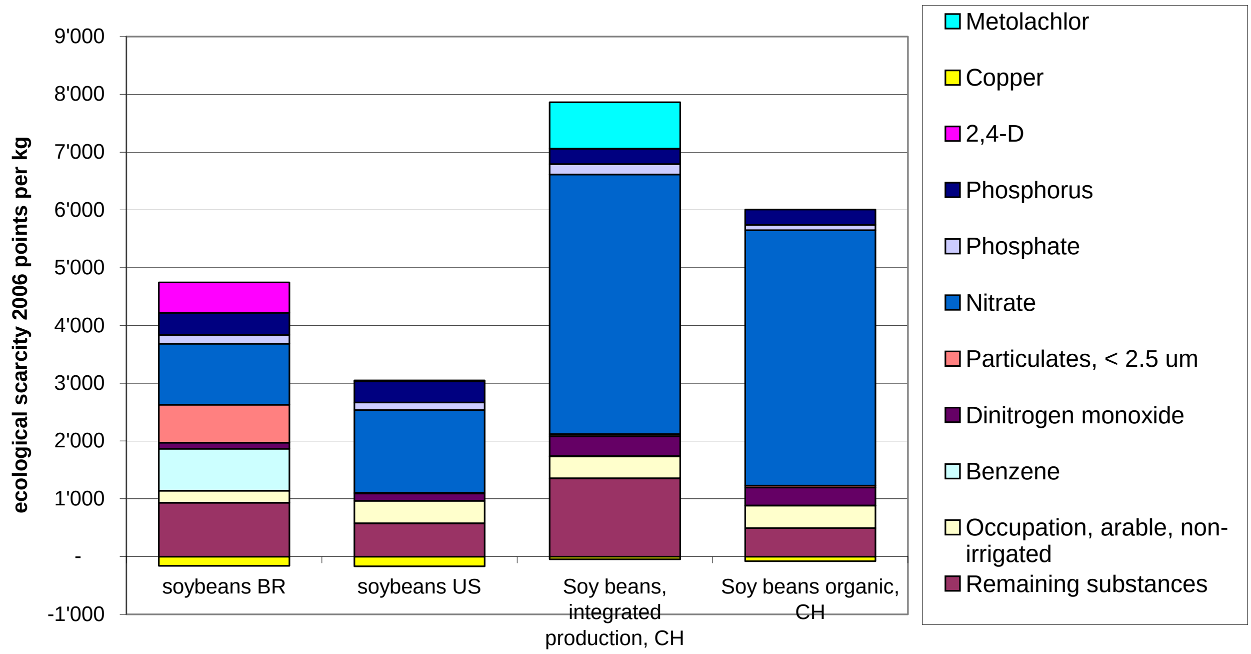


Figure 2 Life cycle impact assessment with the methodology ecological scarcity 2006 of the production of 1 kg soybeans (points per kg)

3.2. Sugar production

A second example shows important aspects of sugar production. In Brazil sugar cane residues are burned to simplify the manual labour of the cutting and harvesting. This leads again to the emission of particles, CO and NMVOC. The evaluation in Figure 3 investigates the respiratory effects with the methodology Eco-indicator 99 (H,A) [5]. The main regions for sugar cane cultivation are located in traditional agricultural areas in the back-country of Sao Paulo and far away from tropical rain forest. Thus, CO_2 emissions from land transformation are not important for the assessment.

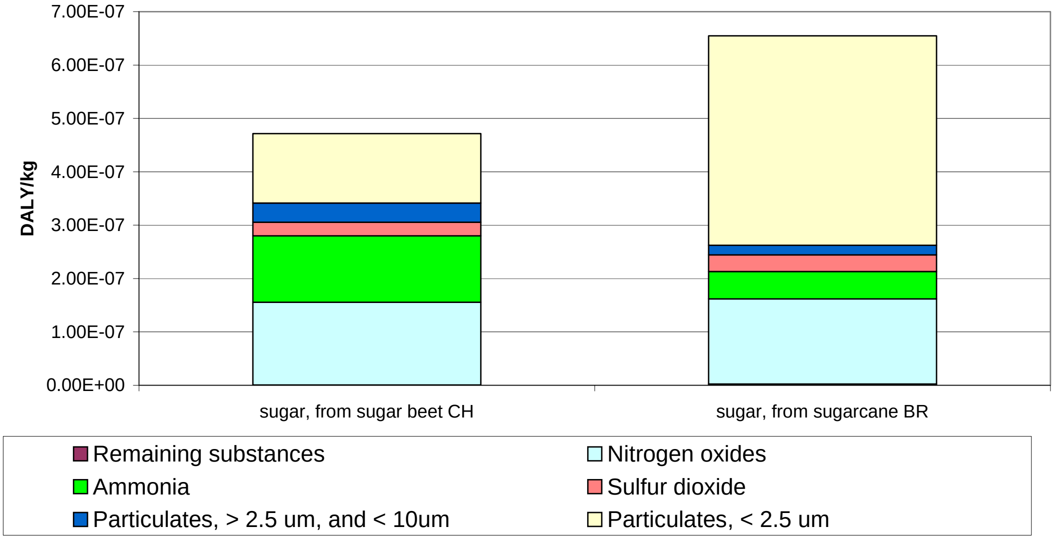


Figure 3 Life cycle impact assessment with the methodology Eco-indicator 99, respiratory effects due to inorganic substances (H,A) of the production of 1 kg sugar (points per kg)

4. CONCLUSIONS

The examples show that agricultural products must be investigated on a regional level. It is quite important to include region specific problems in the LCA. Such problems are e.g. the clear cutting of primary forests and the burning of residues. CO_2-emissions due to land transformation must be considered as an important contributor to global warming in LCA. The ecoinvent data v3.0 provide the necessary information for several agricultural products used for biofuels.

5. REFERENCES

1. Jungbluth N, Chudacoff M, Dauriat A, Dinkel F, et al., *Life Cycle Inventories of Bioenergy*. 2007, Final report ecoinvent No. xx (draft), ESU-services: Uster, CH.
2. Frischknecht R, Jungbluth N, Althaus H-J, Doka G, et al., *Overview and Methodology*. 2004, CD-ROM, Final report ecoinvent 2000 No. 1, Swiss Centre for Life Cycle Inventories: Dübendorf, CH. www.ecoinvent.org.
3. Zah R, Böni H, Gauch M, Hischier R, et al., *Ökobilanzierung von Bioenergie: Ökologische Bewertung*. 2007, Schlussbericht, Entwurf, Abteilung Technologie und Gesellschaft, Empa im Auftrag des Bundesamtes für Energie, des Bundesamtes für Umwelt und des Bundesamtes für Landwirtschaft: St. Gallen.
4. Frischknecht R, Steiner R, & Jungbluth N, *Methode der ökologischen Knappheit - Ökofaktoren 2006*. 2007, Schriftenreihe Umwelt (noch im Entwurf), Bundesamt für Umwelt (BAFU), ÖBU Schweizerische Vereinigung für ökologisch bewusste Unternehmungsführung: Bern.
5. Goedkoop M & Spriensma R, *The Eco-indicator 99: A damage oriented method for life cycle impact assessment*. 1999, Methodology Report, PRé Consultants: Amersfoort, The Netherlands.

Biodiversity impact from agricultural production

An De Schryver[a] and Mark Goedkoop[a]
PRé consultants b.v., Plotterweg 12, Amersfoort 3821 BB, The Netherlands.

Abstract
Agricultural production has a significant impact on the space available for habitats and the biodiversity of an area. Unfortunately, as many impact assessment methods exclude the impact of land use, many food life cycle assessments are carried out without taking into account these effects. In our research we developed new characterisation factors for land use, expressed as the Potential Disappeared Fraction (PDF). We paid special attention at arable areas and distinguished three archetypes of land use intensiveness: monocultures, intensive and extensive areas. Because the availability of edges and borders determines the richness of arable land, we used the species richness in the boundary to calculate the PDF.
For our calculations, we used data of the work of Koellner (2006), the Countryside Survey 2000 (UK), and Crawley and Harral (2001). We took into account the species area relationship, with a different z for each different land use type. Finally, our results are implemented in the ReCiPe method.

1. Introduction

Land use, together with acidification and eutrophication, is known as an important form of pressure affecting biodiversity. For example, only 14.7% of the total terrestrial area of the Netherlands is defined as nature. The rest is used for agriculture and buildings. As a result, the impact of land cover changes on ecosystems is a topic of wide interest in LCA. Within the ReCiPe project[1] special attention is paid to updating and expanding the land use method used in Eco-indicator 99. This paper describes the calculation of endpoint characterisation factors for loss of species diversity, caused by land use occupation. The endpoint characterisation factor for land use is based on the Potential Disappeared Fraction (PDF) of species. This represents the fraction of plant species lost or gained, in relation to a reference. The potential disappeared fraction is influenced by the area-species relationship, also called the island biogeographical theory of McArthur and Wilson (1967). This relationship describes the rising number of species present due to a rising area size, and can be presented as follows:

$$S = cA^z \tag{1}$$

in which S represents the number of plant species, A is the size of the area (m^2), c stands for the species richness factor and z is the species accumulation factor. The factors c and z are specific for each land use type, while factor z is also dependents on the area size. The work of Crawley and Harral (2001) is used to determine the species accumulation factor z. The species richness factors c is calculated using the work of Koellner and Scholz (2006) and the Countryside Survey 2000 of the UK.
To make a proper decision about the reference area, we looked at the potential land use types in Europe, which would appear without any human intervention. According to Stanners and Bourdeau (1995), 80-90% of Europe would be covered by forest. Based on this, the reference area is chosen to be “woodland”.

[1] The objectives of the ReCiPe project is to harmonize two life cycle impact assessment (LCIA) methods, CML 2000 and Eco-indicator 99, into a new method and to improve and update the environmental mechanisms used.

2. Regional damage due to occupation

The regional damage describes the marginal species loss in the surrounded area, due to the fact that occupation reduces the size of the surrounding area and thus the number of species found in that area.

$$CF_{occ} = \frac{\Delta S_R}{S_R} \qquad (2)$$

With CF representing the Characterization Factor, S_R the species number in the region and ΔS_R the difference between natural and current number of species.
The marginal species loss is multiplied by area and time to get the damage caused by the occupation:

$$ED_{occ} = \frac{\Delta S_R}{S_R} \cdot t \cdot A_R \qquad (3)$$

With ED representing the Environmental Damage, A_R the area size of the reference area (m^2) and t the time the area is occupied (year).

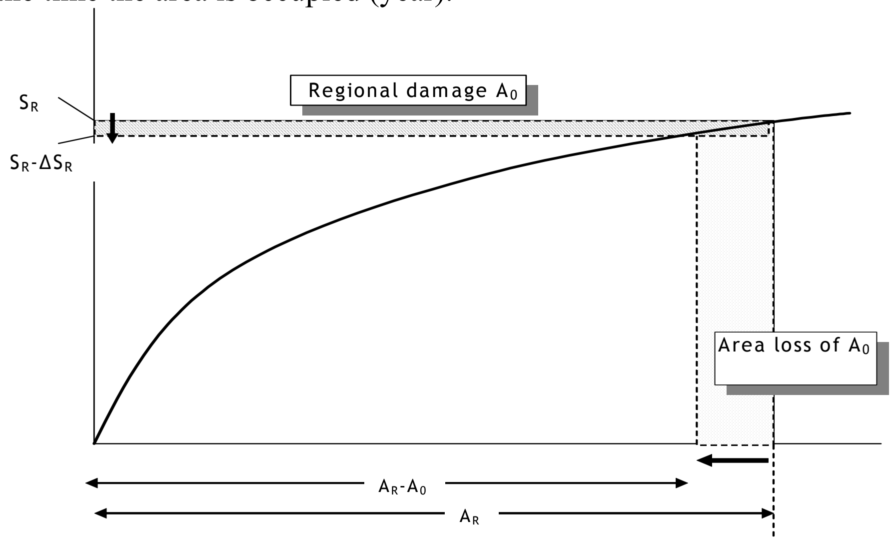


Figure 1: The regional damage for region A_R can be represented as the darker area in the top of the figure

Using formula (3) and the first derivative of formula (1), we get the regional damage factor:

$$ED_{occ-A}(regional_{assumptionA}) = t.A_R.A_0 \frac{z_R c_R A_R^{zR-1}}{c_R A_R^{z}} = z_R \cdot A_0 \cdot t \qquad (4)$$

3. Local damage due to occupation

If the area was not occupied, we would find the number of species on that area using the species area relationship of the reference area. The species number found on the area A_0 before occupation is:

$$S_{occ}(local) = c_R A_0^{z_R} \qquad (5)$$

After occupation, we can expect a number of species (in the occupied area A_0) that is characterised by:

$$S_{i\_local} = c_i A_0^{z_i} \qquad (6)$$

Using formula (5) and (6), we can now determine the species loss on the area A_0:

$$\Delta S_{local} = c_R A_0^{z_R} - c_i A_0^{z_i} \qquad (7)$$

The local environmental damage in area A_0 can be summarized as:

$$ED_{occ}(local) = t.A_0 \frac{S_{R\_local} - S_{i\_local}}{S_{R\_local}} = t.A_0 \frac{c_R A_0^{z_R} - p_i A_0^{z_i}}{c_R A_0^{z_R}} = \frac{c_R - c_i A_0^{z_i - z_R}}{c_R} \cdot A_0 \cdot t \qquad (8)$$

This means, the local characterisation factor depends on the area size of occupation. Because in the LCI data practitioners do not know the occupied area size, a practical solution is needed and will be described underneath.

4. Determination of the species accumulation factor z

According to several sources, the value of the species accumulation factor z in formula (1) can vary between 0.1 and 0.4, depending of the size of the area and the type of land. A further complication in our methodology is that the species accumulation factor z is also dependent on the size of the area.
In our calculations, we used different z factors for different land use types, determined by M.J Crawley (2001). Further, in order to investigate the area dependency of z and formula (8), we calculated the local ecosystem damage using formula (8), for a range of different area sizes. Therefore, we combined the different species accumulation factors of Crawley and Harral (2001) and the species richness factors c of Koellner (2006) and from the data Countryside Survey 2000. Only land use types which have a sufficiently stable characterization factor are selected as usable and an area size of 10,000m^2 is chosen as most realistic to be used in formula (8).

5. Boundary effects in agricultural area's

In our approach, we consider the boundaries of an agricultural field as part of that field. Based on that decision, the species richness of the border determines the species richness of the total agricultural field. The Countryside Survey CS2000 gives species counts in arable land use types for three kinds of plots in Great Brittan (see figure2).

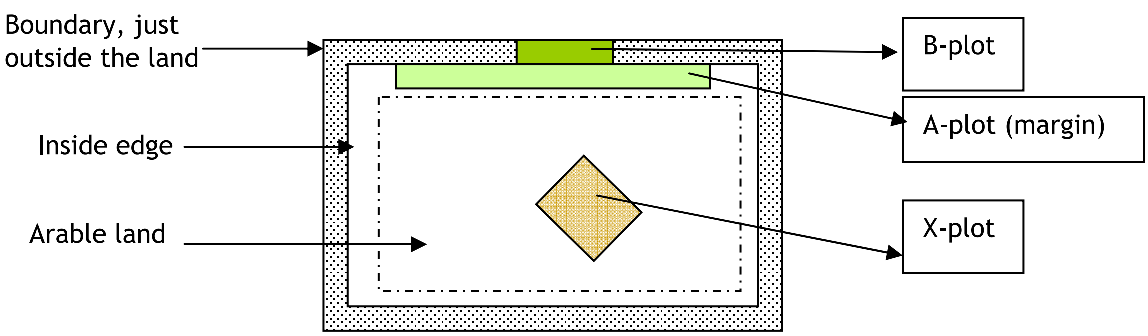


Figure 2: Three kinds of plots used in the country side survey 2000. (Not to scale)

We propose to use three archetypes of land use intensiveness:
1. For monocultures we assume that the species diversity in the entire area is represented by the diversity found in the X plots.
2. For intensive used arable areas we assume that the species diversity in the entire area is represented by the diversity found in the Margins, the A plots.
3. For extensive used arable area's we assume that the species diversity in the entire area is represented by the diversity found in the boundary area's just outside the land, the B plots.

6. Conclusions

We produced characterisation factors for 11 different land use types. The characterisation factors are produced on an endpoint level and based on potential disappeared fractions of species. We used different z-factors for different land use types and distinguished three types of land use intensiveness.

7. References

M.J. Crawley and Harral, J.E., 2001. Scale dependence in plant biodiversity. Science, volume 291, p 264-268.

D. Stanners and Philipe, B., 1995. Europe's environment. The Dobris Assessment. European Environment Agency, Denmark. ISBN 92-826-5409-5.

Countryside Survey 2000: Survey of Broad Habitats and Landscape features ISBN: 1 85112 460 8.

Koellner T and R.W. Scholz (2006): Assessment of land use impacts on the natural environment. Part 2: Generic characterisation factors for local species diversity in central Europe. Int J LCA, online first <DOI: http://dx.doi.org/10.1065/lca2006.12.292.2>

Bastian and Schreiber, 1999. Analyse und ökologische Bewertung der Landschaft. 2nd ed. Heidelberg: Spektrum. P298.

Van Assalt M. and Rotmans J., 1995. Uncertainty in integrated assessment modelling, a cultural perspective based approach. RIVM Report no. 461502009.

Hofstetter P., 1998. Perspectives In Life Cycle Impact Assessment. A Structured Approach To Combine Models Of The Technosphere, Ecosphere And Valuesphere. Kluwer Academic Publishers.

ReCiPe2007: report is under development.

How to account for emissions from manure? Who bears the burden?

Randi Dalgaard and Niels Halberg

Faculty of Agricultural Sciences, Department of Agroecology and Environment, University of Århus, P.O. Box 50, 8830 Tjele, Denmark

1. BACKGROUND

Manure affects the environment negatively because it causes emissions of ammonia, nitrous oxide, nitrate and phosphate, both during storage and when the manure is applied as fertilizer to field-grown crops. On the other hand manure might also contribute positively to the environment, if it substitutes artificial fertilizer or is used for energy production and thereby substitutes fossil fuel. In an integrated farming system where manure is recycled to feed crops only, it does not matter whether manure emissions are allocated to the pigs or the feed crops, since the environmental burden will be allocated to the pigs in any case. But when manure is used in cash crop production, whether on the pig farm itself or after export to another farm, then the question of allocation of emissions from handling manure arises. In order to facilitate comparisons of LCAs on food items it is important to have clear and transparent methods, and –ideally- to agree on a standard method. The following paper will present a practical example with demonstration of the method applied in the Danish LCAfood database (www.LCAfood.dk).

2. OBJECTIVE

The objective is to establish a framework for handling livestock manure in LCA, and thereby give answers to the following question: How to account for emissions from manure in an LCA of livestock products? Shall the environmental impact from manure be ascribed to the pig or the cash crops to which the manure is applied?

3. METHODOLOGY

Our conceptual choice is that all extra emissions arising from using livestock manure in cash crop production should “burden” the environmental profile of the livestock products. On the other hand, this environmental cost should be deducted any saved emissions arising in the cash crop production from replaced fertiliser. Thus, we follow principles of using systems expansion for handling of co-products in LCA [1].
Consequential LCA modelling was performed, thus including the manure related emissions on the cash crop farm and the avoided production of artificial fertilizer. Calculation of the emissions from stable, storage and filed was based on Dalgaard et al. (2006) [2]. The amount of avoided artificial fertilizer is based on data from the Danish Environmental regulation. The Danish regulation stipulated that for each 100 kg of N applied in pigmanure to a crop the fertiliser should be reduced by 60 kg N compared to the public norm for the particular crop on the particular soil type.

The second methodological choice was that if the manure was used for biogas production, the net benefit in terms of avoided CO_2 emissions –and any other avoided emissions- were deducted from the environmental assessment of the pig products.

4. RESULTS AND DISCUSSION

The inventory and characterized results per kg manure N exported from a pig farm to a cash crop farm is presented in figure 1. Each kg manure N exported from the farm results in an avoided production of 0.6 kg N artificial fertilizer, and extra emissions of N and fossil CO_2. Using manure on cash crops instead of fertiliser in cash crops creates more emissions of Nitrogen (ammonia, nitrous oxide and nitrate) contributing to several environmental impact categories. It does not seem satisfactory to leave this as an extra burden on the cash crops. The method presented takes as a starting point that these emissions should burden the livestock products, but only after a proper systems expansion model has been established. The paper has presented how this may be done relatively easy. Due to the strict and detailed Danish regulations for the proportion of fertiliser N to be replaced by manure N there was a transparent reference for calculation of the avoided CO_2 and N emissions from saved fertiliser. In countries where this is not the case there is a need to develop an approach building on representative data re. the degree of fertiliser replacement from manure in the farming systems in question.

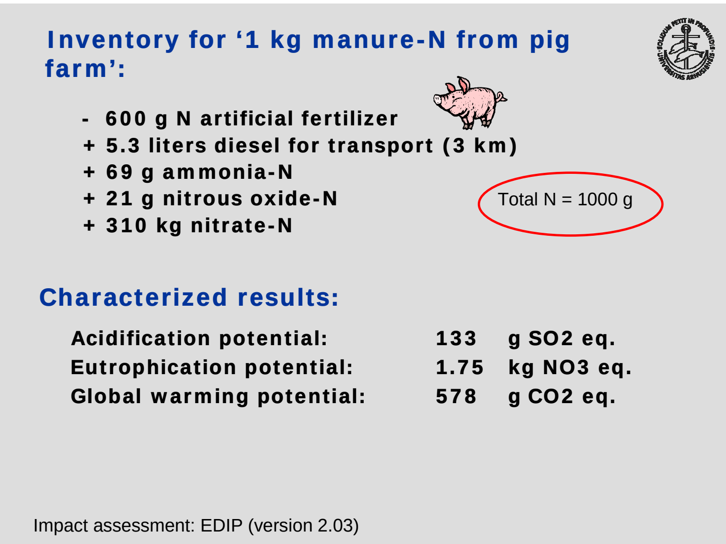


Figure 1. Inventory and characterized results of one kg manure-N exported from a pig farm to a cash crop farm. under Danish conditions.

5. CONCLUSION

The method is easy to apply and gives a coherent methodological alternative to simple (or no) allocation. Both the drawbacks (emissions from stable, storage, fields, transport) and the benefits (e.g. avoided production of artificial fertilizer and fossil energy) must be included. The pig bears the burden from the manure related emissions on the cash crop farm, but the pig also benefits from avoided production of artificial fertilizer and fossil energy.

6. REFERENCES

1. Weidema B (2003): Market information in Life Cycle assessment. Environmental Project No. 863. Danish Ministry of the Environment. Environmental Protection Agency. Available at: http://www.mst.dk/homepage/default.asp?Sub=http://www.mst.dk/udgiv/publications/2003/87-7972-991-6/html/default_eng.htm

2. Dalgaard, N. Halberg, I.S. Kristensen and I. Larsen
Modelling representative and coherent Danish farm types based on farm accountancy data for use in environmental assessments**.** Agriculture, Ecosystems & Environment, 117: **223-237, 200**

Consequential and attributional LCA of conventional and organic milk production

M.A. Thomassen[a], R. Dalgaard[b], R. Heijungs[c], I.J.M. de Boer[a]

[a]Animal Production Systems Group, Wageningen University and Research Centre, PO Box 338, 6700 AH Wageningen, the Netherlands
[b]Faculty of Agricultural Sciences, Department of Agroecology and Environment, University of Århus, PO Box 50, 8830 Tjele, Denmark
[c] Institute of environmental sciences (CML), Leiden University, Leiden, the Netherlands

Abstract
Different methodologies within LCA are used to compare milk production systems. A need exists to evaluate the effect of these choices within LCA. The main goal of this study was to evaluate the effect of using attributional or consequential LCA on the comparison between an average conventional and organic milk production system in the Netherlands. Attributional LCA (ALCA) describes the pollution and resource flows within a chosen system attributed to the delivery of a specified amount of the functional unit. Within ALCA mass and economic allocation were applied. Consequential LCA (CLCA) estimates how pollution and resource flows within a system change in response to a change in demand for the considered product. Within CLCA allocation was avoided by using system expansion. Results showed similar outcomes when comparing ALCA and CLCA on land use (organic higher), energy use (conventional higher), eutrophication (conventional higher) and acidification (conventional higher). Concerning climate change, however, CLCA showed the organic system was higher, while ALCA showed no difference between the systems. CLCA and ALCA are both relevant but answer different questions. This study showed that different modelling when comparing production systems might result in different conclusions. LCA researchers must know what question they want to answer and accordingly choose methodology with precaution.

Keywords: attributional LCA, consequential LCA, conventional, organic, milk production, allocation, system expansion

Background
In the Netherlands over the period 2001-2005, on average 10.7 million ton milk was produced annually [1]. This milk was produced mostly on specialised dairy farms (24.400 in 2001 and 20.810 farms in 2005) that used inputs, like concentrates and artificial fertiliser [1]. Production of milk causes environmental side-effects, such as emission of greenhouse gases and nutrient enrichment of surface water. The Dutch society pays much attention to ecological sustainability of milk production [2]. Organic milk production seems one way to diminish the environmental impact [3]. Life Cycle Assessment (LCA) is identified to be a useful tool to assess the integral environmental impact of different milk production systems, and therefore, was used regularly to compare the integral environmental impact of conventional and organic milk production systems [3-5]. Although guidelines are given how to perform an LCA, still differences among studies exist due to methodological choices. Two LCA approaches, attributional and consequential LCA, were identified and described [6-10]. Attributional LCA (ALCA) describes the pollution and resource flows within a chosen system attributed to the delivery of a specified amount of the functional unit [11]. Consequential LCA (CLCA) estimates how pollution and resource flows within a system change in response to a change in output of the functional units [11, 12].When performing an LCA, in most cases multifunctional processes are included in the analysed system. Choices how to handle co-

products, therefore, are indissoluble connected with performing an LCA. Within ALCA co-product allocation is used most frequently [10]. Avoiding allocation by system expansion is the only way to deal with co-products within CLCA, because it reflects the changes due to a change in production [10]. Usually LCA practitioners choose between ALCA and CLCA to assess the integral environmental impact of a product. This choice affects results of environmental analyses of agricultural products, besides the choice of how to handle co-products. Insight is needed in the effect of choice between ALCA and CLCA and the connected choice how to handle co-products, on the comparisons of LCA results between organic and conventional milk production. The main goal of this study, was to evaluate the effect of ALCA and CLCA on the comparison between an average conventional and organic milk production system in the Netherlands.

Life Cycle Inventory
Table 1 shows the set-up of the milk production systems based on data of 286 conventional and 34 organic farms in the Netherlands of the year 2003 [1]. The system was simplified by assuming that farms were partly self-sufficient.

Table 1 Characteristics of the average conventional and organic milk production system in the Netherlands (reference year 2003) [1, 13]

Characteristic		Unit	Conventional	Organic
Farm area		*ha*	38.5	46.9
Milking cows		*amount*	63	55
Electricity use		*kWh/farm*	25690	26100
Milk production		*kg/cow/yr*	7630	6390
Fat		*%*	4.42	4.40
Protein		*%*	3.49	3.45
Pesticides		**kg/ha*	0.25	-
Concentrates		*kg/cow*	2160	1280
-Attributional	**90 DVE	*ton*	85	58
	120 DVE	*ton*	43	1
	180 DVE	*ton*	7	10
-Consequential	Soybean meal	*ton DM*	71	46
	Spring barley	*ton DM*	64	24

*Active substance matter
** DVE=Dutch system that represents the intestine digestible protein content

Within ALCA, the considered inputs were: natural gas, electricity (based on a power mix), diesel, milk powder, pesticides, artificial fertiliser and concentrates. Within ALCA, purchased concentrates were distinguished in three groups according to their protein and energy content [13]. For each group, a different composition based on national data from the feed industry was used [14]. Within CLCA, the considered marginal inputs were: natural gas, electricity (based on a gas power plant), diesel, milk powder, pesticides, artificial fertiliser, spring barley and soybean meal. Within CLCA, purchased concentrates were related to the identified marginal fodder protein and marginal energy rich component; soybean meal and spring barley [10, 15]. Data on material inputs, like electricity mix, natural gas, fuel oil, fertilisers, pesticides, transport (truck and freighter oceanic) were taken from the Ecoinvent Centre [16]. Capital goods were excluded in both ALCA and CLCA.Within ALCA, economic and mass allocation were applied. CLCA reflects the possible future environmental impact from a change in demand of the product under study. Marginal data were used, which means data representing technologies that are expected to be affected most (the most sensitive) by the chosen change in demand [17]. System expansion was applied. When identifying the marginal avoided burden of meat from dairy cows, for example, the question to be asked was: what will not be purchased by retailers/supermarkets when more meat from dairy cows is provided? The replaced meat was identified to be meat from foreign dairy cows and pork, as meat from dairy cows is mostly used as minced meat and easy-to-prepare meat meals. Meat from foreign and domestic dairy cows, however, is constrained by quotas, and therefore, the marginal meat must come from beef cattle and pigs [10, 18].

Life Cycle Impact Assessment

We used the EDIP 97 method (updated version 2.3) that has been implemented in the PC-tool SimaPro 7.0 [19]. We used the method Cumulative Energy Demand (CED) to assess energy use [20].

Results

Table 3 shows the (characterized) results of both ALCA and CLCA. When comparing conventional with organic milk production, both ALCA and CLCA show organic milk production had a higher land occupation, conventional milk production had a higher fossil energy use, higher eutrophication and higher acidification. Concerning climate change, however, CLCA showed organic milk production was higher, while ALCA showed no difference between conventional and organic milk production.

Table 3 Characterized results expressed per kg Fat and Protein Corrected Milk (FPCM) of conventional and organic milk production assessed by attributional LCA (ALCA) using both mass and, economic allocation, and by consequential LCA (CLCA) using system expansion (se)

Milk production		Conventional			Organic		
Methodology		ALCA	ALCA	CLCA	ALCA	ALCA	CLCA
Handling co-products		mass	eco	se	mass	eco	se
Impact category	*Unit/kg FPCM*						
Land occupation	*m^2*	1.20	1.18	0.85	1.72	1.60	1.44
Fossil energy use^{CED}	*MJ-eq*	5.58	6.75	2.49	2.95	2.84	2.02
Eutrophication^{EDIP97}	*g NO_3-eq*	163	170	113	110	105	56
Acidification ^{EDIP97}	*g SO_2-eq*	10.9	11.2	4.74	8.33	7.93	3.24
Climate change ^{EDIP97}	*g CO_2-eq*	1550	1590	896	1610	1520	1260

Discussion and conclusions

This study showed that different modelling when comparing production systems might result in different conclusions. The higher outcome of climate change of organic milk production by using CLCA was due to higher methane and nitrous oxide emissions at farm level and inputs of the organic system and less avoided burden of the beef and pork production compared with the other impact categories. ALCA and CLCA are both relevant but seem to answer different questions [10]. Most CLCAs were applied when asking change-oriented questions, i.e., what would have happened if (retrospective) or what will happen if (prospective). Most ALCAs were applied for identification of a hotspot or contribution analyses, i.e. what were the most important elements in integral environmental impact during the production of a product? In case of comparing different production systems, LCA researchers must know what question they want to answer and accordingly choose methodology with precaution. We recommend LCA practitioners of different research areas to perform similar case studies to address differences between CLCA and ALCA of the specific products, because outcomes might differ from our study.

References

1. BINternet, 2003. Bedrijven-InformatieNet. In: Landbouwkundig Economisch Instituut, Wageningen, pp. http://www.lei.wur.nl/NL/statistieken/Binternet.
2. van Calker, K.J., 2005. Sustainability of Dutch dairy farming; A modelling approach. Business Economics, Department of Social Sciences. Wageningen University, Wageningen, pp. 1-207.
3. de Boer, I.J.M., 2003. Environmental impact assessment of conventional and organic milk production. Livestock Production Science 80, 69-77.
4. Cederberg, C., Flysjö, A., 2004. Life Cycle Inventory of 23 dairy farms in South-Western Sweden. In: 728, S.-r.N. (Ed.), The Swedish Institute for food and biotechnology, pp. 1-59.
5. Thomassen, M.A., Van Calker, K.J., Smits, M.C.J., Iepema, G.L., De Boer, I.J.M., Submitted. Life Cycle Assessment of conventional and organic milk production in the Netherlands. Agricultural Systems.
6. Heijungs, R., 1997. Economic Drama and the Environmental Stage. Formal Derivation of Algorithmic Tools for Environmental Analysis and Decision-Support from a Unified Epistemological Principle. Centre of Environmental Science. Leiden University, Leiden.
7. Frischknecht, R., 1998. Life Cycle Inventory Analysis for decision-making. In: Swiss Federal Institute of Technology. Zurich, Switserland.
8. Ekvall, T., 1999. System expansion and allocation in Life Cycle Assessment; with implications for wastepaper management. Department of technical environmental planning. Chalmers University of Technology, Goteborg, Sweden, pp. 1-54.
9. Tillman, A.-M., 2000. Significance of decision-making for LCA methodology. Environmental Impact Assessment Review 20, 113-123.
10. Weidema, B.P., 2003. Market information in life cycle assessment. Danish Environment Protection Agency, pp. 1-147.
11. Rebitzer, G., Ekvall, T., Frischknecht, R., Hunkeler, D., Norris, G., Rydberg, T., Schmidt, W.P., Suh, S., Weidema, B.P., Pennington, D.W., 2004. Life cycle assessment: Part 1: Framework, goal and scope definition, inventory analysis, and applications. Environment International 30, 701-720.
12. Ekvall, T., Weidema, B.P., 2004. System boundaries and input data in consequential life cycle inventory analysis. Int J Life Cycle Assessment 9, 161-171.
13. ter Veer, D.F., 2005. Scenariostudie 100% biologisch voeren melkvee. Animal Sciences Group/Praktijkonderzoek, Lelystad, pp. 1-22.
14. Doppenberg, J., de Groot, J.P., 2005. Lineaire programmering rundvee-, varkens- en pluimveevoeders. VVM Bedrijfsbureau; Vereniging Voorlichting Mengvoerindustrie, Deventer, The Netherlands.
15. Dalgaard, R., Schmidt, J., Halberg, N., Christensen, P., Thrane, M., Pengue, W.A., Submitted. Consequential LCA of soybean meal. International Journal of Life Cycle Assessment.
16. EcoinventCentre, 2004. Ecoinvent data v1.1 Final reports ecoinvent 2000 (1-15). Swiss Centre for Life Cycle Inventories, Dübendorf, pp. CD-ROM.
17. Schmidt, J.H., Weidema, B.P., Submitted. Shift in the marginal vegetable oil. International Journal of Life Cycle Assessment, 1-6.
18. Weidema, B.P., 2006. Answer to questions concerning consequential LCA milk, Enveco course, Ålborg,. Thomassen, M.A. Wageningen, the Netherlands
19. Wenzel, H., Hauschild, M., Alting, L., 1997. Environmental assessments of products- Vol 1:Methodology, tools and case studies in product developments. Edition, F. (Ed.), Champan&Hall, London (UK); Weinheim (Germany); New York (USA); Tokyo (Japan); Melbourne (Australia); Madras (India)
20. VDI, 1997. Cumulative Energy Demand - Terms, Definitions, Methods of Calculation. Ingenieure, V.D. (Ed.), VDI-Richtlinien 4600. Düsseldorf, Germany.

Improved Sustainability of Agro-Food Chains in Central America - How to use socioeconomic indicators (SEI) as complement to LCA

Anna Flysjö[a], Thomas Ohlsson[a], Roldan Muradian[b] and Wim Pelupessy[b]

[a] Department of Environment and Process Engineering, The Swedish Institute for Food and Biotechnology (SIK), Box 5401, 402 29 Gothenburg, Sweden
[b] Development Research Institute (IVO), Tilburg University, P.O. Box 90153, 5000 LE Tilburg, The Netherlands

Abstract

There is an increasing concern among European consumers about the impact of their consumption habits. In the last decennium the focus has primarily been on the environmental issues, with a growing interest for organic foods and for various labelling initiatives for foods produced by environmentally improved methods. In the last number of years the focus has shifted towards inclusion of the other aspects of sustainability – the social and economical conditions in the production system.

Within the project 'Improved Sustainability of Agro-Food Chains in Central America' the production and trade of different global commodity chains have been investigated. The objective of the project was to identify technological options to make agro-food systems more sustainable from an environmental, food safety, and socio-economic perspective, using a market-oriented approach. In the understanding of sustainability of food chains the environmental aspects of resource use and emissions were combined with considerations on the impact on society and on the socio-economical conditions for the actors in the food chain.

This presentation focus on how the socio-economic indicators (SEI) have been used as a complement to lifecycle assessment (LCA) for different agro-food chains in Central America, the selection and development of indicators (SEI) and challenges with the integration of SEIs into the LCA methodology. Finally, selected results will be presented for snow peas produced in Guatemala.

Introduction

The European consumers are increasingly not only concerned about the product quality but also the production quality of the food they eat. In the last decennium the focus has been on the environmental issues and in the last number of years the focus has shifted to also include the other aspects of sustainability, the social and economical conditions of the entire production system or chain. In the understanding of sustainability of food chains the environmental aspects of resource use and emissions are combined with considerations on the impact on society and on the socio-economical conditions for the many actors in the food chain.

This paper is based on the results of a study on extending LCA to include also socio-economic indicators performed in an EU INCO (International Scientific Cooperation) project with the title: Improved Sustainability of Agro-Food Chains in Central America, (ICA4-CT-2002-10010). The aim of the project was to contribute to more sustainable food production systems in Central America focusing on coffee, fruits, nuts and vegetables and to identify technological options that make agro-food systems more sustainable from an environmental, food safety and socio-economic perspective. Special attention was given to the position of small agro-food producers in developing economies and their potential to generate higher value added products. The project intends to develop technology evaluation tools for sustainable development which will be generally applicable to agro-food chains between developing and developed countries. This should lead to results that are directly applicable, leading to the interactive assessment of exploitable technological options for important market-oriented agro-food products in Central America. The products studies are significant valued added and foreign exchange generators, mainly produced by smallholders and with considerable demand potential in the European Union.

In this paper the development of the method, to assess socio-economic indicators (SEI) along with environmental aspects, and selection of indicators will be presented, together with some practical examples of the results from one of the case studies: production of snow peas in Guatemala.

Method and results

As life cycle assessment (LCA) is a well established method based on a system analysis approach for assessing environmental impacts, it was decided early in the project to base the development of the method for sustainability indictors on the LCA method. In the selection of SEIs a first step was to study previous work and also at different international agreements and conventions. It is important to have socially relevant indicators and therefore the choice of socio-economic indicators should preferably be based on international codes of conduct, which is also in line with the ISO standard that says that "the impact categories, category indicators and characterization models should be internationally accepted i.e. based on and international agreement or approved by a competent international body" (ISO. 2006). Already in an early stage of the project it was realised that it was preferable to have a rather limited set of indicators to make the inventory a feasible task and a set of key indicators was selected for this study (see list of selected indicators presented under results in table 1).

Table 1: List of the selected socio-economic indicators

Indicator	Description	Unit
Value added along the chain	how is the value distributed along the chain	USD per kg
Fair price	do the producers get a "fair" price for their products	USD per kg
Costs	how large are the costs of production	USD per kg
Income	is the salary enough to cover "basic needs"	USD per day
Working hours	how many hours per day do the workers work	hours per day
Legal contracts	do the workers have legal contracts	% of workers
Access to health care	do the workers have access to health care	% of workers
Level of education	what is the level of education at the farm	% of workers
Gender	how many of the workers are female/male	% of workers
Migrants	how many of the workers are migrants/non-migrants	% of workers
Child labour	how many of the workers are below fifteen years old	% of workers
Use of chemicals	what type and amounts of chemicals are used amounts	kg per ha

There are some fundamental differences between the 'traditional' LCA method and SEIs. Traditional LCA is based on biophysical flows, which are easy to quantify and the results are relatively easy to compare between different systems, since they all can relate to a functional unit. SEIs on the other hand have not this connection to a functional unit. Rather than be product (or process) related, as traditional LCA usually is, the SEIs are often more related to the performance of the company, as explained in Dreyer et. al. 2006. This makes it difficult to integrate socioeconomic aspects directly into the LCA framework. Though, this might be feasible if all indicators would be possible to quantify. Suggested methods for integrating social and economic aspects into the LCA framework, by connecting indicators by impact pathways to for example damage categories (some midpoint or endpoint value), is also described in Weidema (2006) and Norris (2006).

In this paper the chosen indicators is used in addition to the LCA results and no attempts to creating pathways to different damage categories is done. Instead a set of key indicators, presented in table 1, have been selected to in a simplified way illustrate the socioeconomic situation for production of agro-food products in developing countries, more specifically Central America. The list represents both indicators important for the worker (salary, working conditions, legal contracts and access to health care) as well for the owner/employer (price of product, costs of production, profit) and it also gives a brief description of the workforce (gender, child labour, immigrants, level of education). Of course there would be more indicators of importance, but there is a need for limitations and all the indicators presented here was agreed to be important.

During the actual inventory of the indicators at the farms, the field work gave important information on the feasibility of getting informed answers to the questions indicated on the spreadsheets. There were a number of reasons for difficulties of collecting the desired information, such as insufficient book keeping, sensitive information from a commercial point of view or difficulties in personal relations between employer and employee.

Within the project several products were studied, but here only some results from the case study on snow peas in Guatemala are used as an example on how the socio-economic indicators could be used.

Snow peas in Guatemala

The case study presented here compare snow peas produced for the US market with snow peas produced for the EU market (Espindola Rafael, 2006). For both options the snow peas were produced in Guatemala and the main focus of the results are on farm level, since the focus of the overall project were to analyse the situations for farmers. Figure 1 shows the percentage of the consumer price, 5.43 and 5.68 USD per kg, of snow peas for the market of US and EU respectively. There is a large difference in the percentages which can be explained by the difference in transportation; the price of transportation (export harbour to import harbour), which is longer to EU and therefore presumably higher, is included in the exporter, while the transportation to retail (included in retail) might be longer in the US and therefore the higher percentage. The farmer though, only gets a minor piece of the consumer price; 11.5 and 14.0 percent for US and EU respectively.

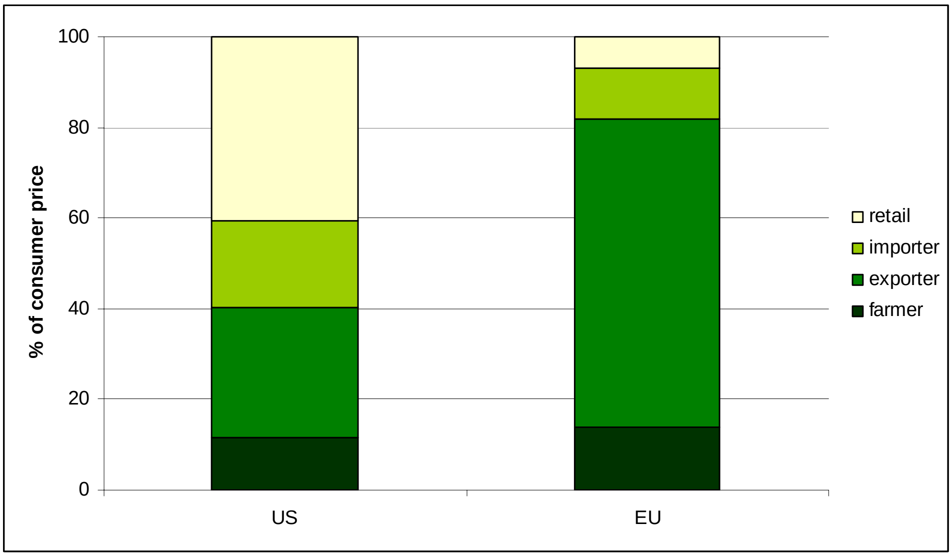


Figure 1: Percentage of consumer price for snow peas exported to the EU and US market respectively.

The costs for different inputs are shown in figure 2. There is a higher cost for agrochemicals and fertilisers for the snow peas produced for the US market, while the labour costs is higher in the case for the EU market.

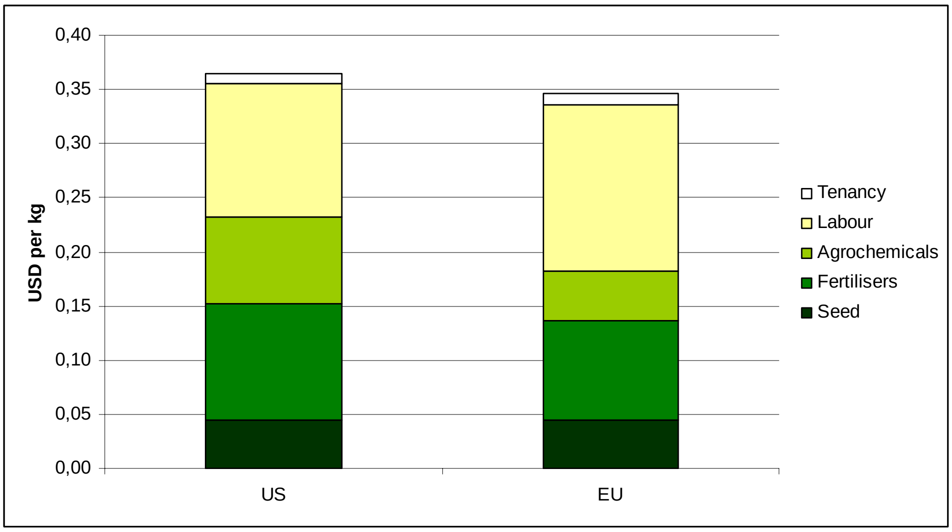


Figure 2: Cost of inputs for farmers exporting to the EU and US market respectively.

The producers for the EU market have more hired workers while the producers for the US market have more family workers, which explains the lower labour costs for farmers exporting to US, though they have a higher input of working time per kg of product along with lower yields. In total the producers for the US market have 25 percentages more working time per kg of product compared with the producers for the EU market. It could also be mentioned that the workers producing for the EU market have an additional 12 percentage higher salary (3.96 USD compared to 3.54 USD per day) than those producing for the US market.

Discussion and recommendations

Some of the more methodological problems with the inclusion of SEIs into the LCA framework were the conceptual differences of the systems. As mentioned, 'traditional' LCA relates to a product or process which is not the case of SEI. SEIs is also more often qualitative as well as quantitative, while inventory data for LCA usually are quantitative (though biodiversity, land use and pesticide use are examples of environmental impacts which are of a more qualitative character, hence there are also a lot of research going on to improve assessments of these impacts within the LCA methodology). Even though the indicators selected here were tried to be quantified, it was still decided to have the indicators more as a "complement" to LCA. Depending on the purpose of the project, it could be easier and better for communication to only use indicators, instead of trying to aggregate them (or fully integrate into the LCA framework).

When performing an LCA several impact categories are usually studied to get an understanding of the "total" environmental impact. In the same way it is important to study other impacts, besides environmental ones, to get an understanding of the sustainability. This might be especially important in developing countries where, for example, manual labour often is used in stead of machinery labour, and working conditions might not be so good. LCA has been shown to be a useful tool for assessing environmental impacts, but to get a representative picture of the sustainability of production, it is important to also include other aspects (i.e. social and economic), which has been shown to be feasible during this project, and more work and research would be desired on this topic, not at least as case studies.

Dreyer, L., Hauschild, M., Schierbeck, J. 2006. A Framework for Social Life Cycle Impact Assessment. Int J LCA 11 (2) 88-97

Espindola Rafael, V. 2006. Assessment of the Guatemalan snow peas chain. Master thesis at Wageningen University, Development Economics Group

ISO. 2006. Environmental Management – Life cycle assessment – Requirements and guidelines. ISO 14044:2006(E). International Organization for Standardization. Geneva. Switzerland

Norris G (2006): Social Impacts in Products Life Cycles Towards Life Cycle Attribute Assessment. Int J LCA 11, Special Issue 1, 97-104

Weidema B (2006): The Integration of Economic and Social Aspects in Life Cycle Impact Assessment. Int J LCA 11, Special Issue 1, 89-96

Socioeconomic Indicators as a Complement to Life Cycle Assessment -An Application to Salmon Production Systems

Sarah A. Kruse[1], Anna Flysjö[2], Nadja Kasperczyk[3], and Astrid J. Scholz[4]

[1] Ecotrust, 721 NW Ninth Ave., Suite 200, Portland, Oregon, USA, skruse@ecotrust.org
[2] The Swedish Institute for Food and Biotechnology (SIK), Box 5401, S-402 29, Göteborg, Sweden, afo@sik.se
[3]The Institute for Rural Development Research (IfLS), Zeppelinallee 31, 60325 Frankfurt am Main, Frankfurt, Germany, kasperczyk@ifls.de
[4] Ecotrust, 721 NW Ninth Ave., Suite 200, Portland, Oregon, USA, astrid@ecotrust.org
***Corresponding author** (skruse@ecotrust.org)

Keywords: LCA; SEI; socioeconomic; sustainability; indicators; salmon; seafood

Abstract

This paper presents the methods for and development of a suite of socioeconomic indicators that complement the LCA methodology and provide a comprehensive approach for assessing the cradle-to grave sustainability of a product or process. A combined top-down and bottom-up approach serves as the basis for development of the set of socioeconomic indicators (SEIs) presented here. Generally recognized societal values, industry specific issues and the financial constraints associated with collection of data necessary for measurement of the indicators are all factors considered in this approach. Indicators are categorized based on fundamental methodological differences and then used to describe the socioeconomic impacts associated with salmon production. There is a need to further develop and refine methods to assess the results of socioeconomic indicators using a life cycle perspective. The SEIs presented here are discussed theoretically within the context of salmon production systems, but in order to test the practicability and validity of the indicators a practical application is also necessary.

Introduction

There is a growing recognition on the part of industry, policymakers and consumers that sustainable industry practices are needed to maintain environmental and social well-being. Even though the life cycle assessment (LCA) methodology has the potential to include both social and economic indicators, and SETAC guidelines recommend the inclusion of such impact categories in all detailed LCAs, no established set of metrics exists to describe the relationship between socioeconomic indicators (SEIs) and a specific product/process, nor is there a common understanding how such metrics might be developed.

To date, there have been only limited attempts to include social aspects in the LCA framework, though the effort has been increasing (Hofstetter et al. 2006, Norris 2006, Weidema 2006, Dreyer et al. 2006, Labuschagne 2006, Hunkeler 2005, Klöpffer 2003, Heller 2000, O'Brien et al. 1996). The methods used are largely inconsistent with one another, however, and the majority of studies have concluded that more research and development is needed in this area.

The focus of this paper is to identify relevant and suitable socioeconomic indicators and show how these could be used as a complement to LCA. Using the example of salmon, this paper illustrates how socioeconomic indicators can provide comparative information to assess the relative impacts associated with comparable products coming from different production systems (e.g. wild salmon fillet versus farmed salmon fillet) that can inform both consumers' personal practices and policymakers' decisions.

Methods

The goal of integrating socioeconomic indicators into the LCA framework requires some discussion of the approach used to identify and select indicators, as well as how indicator definition differs between the traditional framework and a framework that includes socioeconomic indicators. It should be noted that because the focus is on indicator development, the valuation portion of LCA is considered outside the scope of this article.

Similar to Dreyer et al. (2006), we believe that a combined top-down and bottom-up approach must be used in order to develop a defensible suite of indicators. A top-down approach, in this context, is one that selects indicators that are representative of broadly recognized societal values. To the extent possible, indicators are based on various international conventions, agreements and guidelines. In contrast, we also define a bottom-up approach as one that identifies indicators based on industry or stakeholder interests and/or data availability. Socioeconomic impacts have the potential to vary between industries due to the nature of the process/product with which a given industry is involved. It is crucial that any set of socioeconomic indicators used as a complement to LCA be able to adequately address industry specific impacts. Additionally, a bottom-up approach focuses on use of readily available data.

Life cycle assessment traditionally requires that all flows considered are related to a functional unit. In the case of biophysical flows (i.e. raw materials, energy, emissions, etc.) these relationships are generally direct, quantifiable and easy to establish. Describing the causal relationship between a socioeconomic impact and the product in question, however, may not be as straightforward, nor, in some cases, as easily quantifiable. We suggest that multiple measurement methods are needed for SEIs in order for them to accurately describe the relationship between the socioeconomic impact and the product in question as well as to be successfully integrated into the LCA framework. This new approach is best described through a categorization based primarily on the methodological differences between the indicators. In our categorization, socioeconomic indicators fall into two types: *additive indicators* and *descriptive indicators*.

Additive indicators (must) meet two criteria: 1) they can be measured quantitatively, and 2) they relate to the functional unit (i.e. are additive through the chain). As all additive indicators relate to the functional unit, they are methodologically identical to the traditionally defined biophysical indicators. This distinction makes them widely applicable and directly comparable across different life cycle assessments.

A number of widely recognized socioeconomic sustainability concerns, particularly those related to working conditions, are described by indicators that fail to meet the additive indicator criteria and are neither strictly quantitative nor additive along the chain (i.e. related to the functional unit). Under our categorization such indicators, instead of being left outside the framework, will comprise a second category of indicators, descriptive indicators. We suggest that even if impacts cannot be related to a functional unit, they can still capture life cycle thinking, and thus be valuable from a sustainability perspective, by being described at each point in the chain.

Consequently, descriptive indicators meet the following criteria: 1) they can be either quantitatively or qualitatively described and/or measured at each point in the chain, and 2) they cannot be related to the functional unit (i.e. are not additive through the chain). The category of descriptive indicators can be further broken down into two sub-categories, general and specific, based primarily on the perspective or frame of reference being considered. Like additive indicators, descriptive general indicators are meant to describe broad societal values be widely applicable, and to a large extent reflect a top-down approach (i.e. internationally established standards). For example, descriptive general indicators might describe whether or not workers at each point in the value chain are paid a living wage or receive appropriate worker benefits. Descriptive specific indicators, on the other hand, are those indicators that may not be widely applicable, but rather, are focused on the relevant socioeconomic impacts of a specific process or product. In contrast to descriptive general indicators, the comparability of descriptive specific indicators across different life cycle assessments may be limited to comparison with production systems similar to the one in question. For example, a socioeconomic indicator associated with a wild salmon production system might also be relevant for a farmed salmon production system or even another seafood production system, and as such lend itself to comparisons between these systems.

Applying additive indicators to salmon production

As mentioned previously, all additive indicators can be related to the functional unit and therefore are relevant for salmon production systems, but are also relevant and directly comparable across different life cycle assessments. Additive indicators capture more economic aspects than social aspects, though several indirectly account for both. For example, both the costs indicator and the working hours indicator can be described and measured in terms of gender or migrant labor, making it possible to relate differences between different groups (i.e. male and female or migrant and non-migrant) to the functional unit. These indicators are all quantitative and could be expressed in either US dollars or person hours of production. For example, choosing a functional unit corresponding to one kilogram of salmon ready to eat at the consumer stage would give the production cost or person hours of production through the entire value chain in relation to this functional unit.

Applying descriptive general indicators to salmon production

The descriptive general indicators, as explained earlier, are applicable and may be comparable across different life cycle assessments. These indicators generally focus on describing broad societal values related to working conditions (e.g. living wage, employment benefits, hours worked per week, right to organize, forced labor) and the labor force (e.g. age distribution of workers, education level of workers, gender of workers). These indicators capture aspects that the additive indicators fail to address and are the most crucial for measuring social impacts, specifically those related to broadly recognized societal values.

Applying descriptive specific indicators to salmon production

While descriptive general indicators focus primarily on broadly recognized societal values, descriptive specific indicators, as their name implies, are focused on a specific product or process. From a sustainability perspective, this ability to create measurable indicators that describe the wide range of socioeconomic concerns attributable to different industries is critical. For example, the impact of pesticide use on workers may be of great concern in certain types of agriculture production (e.g. coffee) but is not a factor in wild salmon production systems.

The descriptive specific indicators chosen in the context of salmon production systems describe socioeconomic impacts at three different levels: individual, fishery and societal. The contribution to income indicator measures the contribution of salmon production to personal income (i.e. at the individual level). For salmon production systems, we expect that the first point in the value chain is where the highest contribution to personal income will be seen.

The following indicators describe fishery level socioeconomic impacts: fair price, access to fishery, and latent quota. Indicators relating to owner-operator, adjacency, and compliance are meant to describe the salmon production industry on a broader level.

This set of socioeconomic indicators is meant to serve as a complement to the traditional LCA framework and will serve as the starting point for a first attempt (in a forthcoming paper) to assess the overall sustainability of salmon production, including environmental, social and economic aspects, using a life cycle approach.

Discussion and recommendation

There are a variety of issues that need to be addressed when using SEIs as a complement to the LCA framework. Because SEIs are a new concept, relative to the traditional biophysical LCA for which a variety of databases currently exist, much of the data needed to populate the indicators are not readily accessible, or in some cases are not currently being collected. Given this challenge, recommendations that data necessary to describe and measure SEIs be collected may be one inevitable conclusion of our project. There are some sustainability standards for which data collection is likely to be extremely difficult (e.g. forced labor) and in such cases the use of proxy indicators should be considered as an alternative. As suggested by Weidema (2005), average data is a good way to fill in data gaps, but when performing studies on specific products and processes the need for site specific data may be a crucial issue, especially if the geographic or social context is more important than the activity itself. Alternately, the use of appropriate measurement methods that account for regional differences, i.e. of geography, culture, government etc, and/or the relevance of particular indicators for a particular geographic area may be able to minimize the impact of such differences on the overall assessment.

As mentioned previously, the development of a "sustainability LCA" (i.e. including socioeconomic aspects into the traditional LCA framework) is still in its nascent stages, but a rising demand from stakeholders, along with the increasing research/publications on the topic, shows both a need for and interest in a methodological framework that provides a comprehensive measure (i.e. environmental, social and economic) of process/product sustainability using a life cycle perspective.

References

Dreyer L, Hauschild M, Schierbeck J (2006): A framework for social life cycle impact assessment. Int J LCA 11(2), 88-97

Hofstetter P, Madjar M, Ozawa T (2006): Hapiness and Sustainable Consumption Psychological and physical rebound effects

Klöpffer W (2003): Life-cycle based methods for sustainable product development. Int J LCA 8(3), 157-159

Labuschagne, C (2006): Social Indicators for Sustainable Project and Technology Life Cycle Management in the Process Industry. Int J LCA 11 (1), 3-15

Heller, M C and G A Keoleian (2000): Life Cycle-Based Sustainability Indicators for Assessment of the U.S. Food System. Ann Arbor, MI, Center for Sustainable Systems**:** 61

O'Brien M, Doig A, Clift R (1996): Social and environmental life cycle assessment (SELCA): Approach and methodological development. Int J LCA 1(4), 231-237

Hunkeler D, Rebitzer G (2005): The future of life cycle assessment. Int J LCA 10(5), 305-308

Norris G (2006): Social Impacts in Products Life Cycles Towards Life Cycle Attribute Assessment. Int J LCA 11, Special Issue 1, 97-104

Weidema B (2005): ISO 14044 also Applies to Social LCA (Letters to the Editor). Int J LCA 10 (6), 381

Weidema B (2006): The Integration of Economic and Social Aspects in Life Cycle Impact Assessment. Int J LCA 11, Special Issue 1, 89-96

Ratio and additive models for measuring environmental-economic performance: Case studies at field and farm levels

K. Hayashi[a] and T. Nemecek[b]

[a]Environmental Assessment and Management, National Agricultural Research Center, National Agriculture and Food Research Organization, 3-1-1 Kannondai, Tsukuba, Ibaraki 305-8666, Japan
[b]Life Cycle Assessment, Agroscope Reckenholz-Taenikon Research Station ART, Reckenholzstrasse 191, CH-8046 Zürich, Switzerland

Abstract
We present ratio and additive models for measuring environmental-economic performance and apply them for analyzing the relationships between the intensity of agricultural management, yield, and environmental impacts at field and farm levels. A yield function that maps the intensity on yield and an impact function that maps the intensity on the environmental impact are used for defining the two models. In the ratio model, the efficiency is defined as the ratio of the value of an impact function to that of a yield function. In the additive model, a composite function of the inverse of a yield function and an impact function is used for the definition. These models are applied for analyzing the data from field experiments and farm records. The results indicate that there is a variety in the shapes of the functions and that using the both models simultaneously will be useful in assessing agricultural practices.

1. INTRODUCTION

The number of applications of LCA to agricultural production systems has increased (Hayashi et al. 2006). One of the important characteristics in the applications is that the problems are defined as the selection (comparison) of a discrete alternative; the selection problem in decision analytic terminology. However, the rank order of the alternatives (the results of the assessment) is dependent on the choice of functional units, e.g., whether to use area-based or product-based indicators. Therefore, in order to clarify the relationship between management intensity (area-based), yield (area-based), and environmental impacts (area- or product-based), we develop ratio and additive models for measuring environmental-economic performance. The two models are applied to case studies at field and farm levels.

2. MODEL DEVELOPMENT

In this section, after defining yield and impact functions, ratio and additive models for measuring environmental-economic performance are presented.

2.1 Yield functions

A yield function ($y = f(x)$) maps management intensity per unit area (x) onto crop yields per unit area (y). It is also termed the fertilizer response function, which measures the partial factor productivity of the applied fertilizer and is a single input version of a production function (Cassman et al. 2003). This function for explaining fertilization intensity and crop yields can commonly be found in the agronomic and related literature (e.g., Cassman et al. 2003, Brentrup et al. 2004, Charles et al. 2006).

2.2 Impact functions

An impact function ($z_i=g_i(x)$) maps management intensity per unit area (x) onto an environmental impact per unit area (z_i). The most typical indicator of management intensity is the fertilization level. The impact can be impact categories in midpoint (environmental theme) approaches and damage categories in endpoint (damage oriented) approaches. This type of function can be found in the previous LCA applications to agriculture (Brentrup et al. 2004), although the term impact function is not used in these studies. In the case of pesticide application, the function can be related to the LCIA of toxic chemicals, whereas in the case of fertilizer application this function can be related to the uptake efficiency of the applied nutrients (Cassman et al. 2003) because environmental impacts are caused by, for example, nitrate leaching and N_2O emissions.

2.3 The ratio model

The decision concerning the selection of agricultural practices can be made using the ratio model or the additive model. This is applicable to pesticide application. In the ratio model, the decision will be made using efficiency defined as the ratio of economic performance to environmental impacts. In this case, efficiency is defined as the ratio of the environmental impact per unit area to the yields per unit area (z_i/y). The ratio can be understood as the reciprocal of eco-efficiency or ecological efficiency (environmental intensity according to the terminology defined by Huppes & Ishikawa (2005)), which is defined as output or value added per environmental impact added (Schaltegger & Burritt 2000). The fact that the ratio becomes the impact per unit yield (functional unit) implies that this model has been used implicitly in the former product LCA studies on agriculture (Hayashi et al. 2006).

2.4 The additive model

The basic idea underlying the additive model is that the yields (y) can be considered as an indicator of economic performance, and the impact (z_i) as an indicator of environmental quality. Thus, using a yield and an impact function, we can construct a two-objective model; one objective is to achieve a good economic performance (to maximize the yield) and the other is to achieve a good environmental performance (to minimize the environmental impacts).

The frontier, depicting the possible combinations of economic performance (y) and environmental quality (z_i), is known as the impact-yield function. In order to determine the optimum point, additional information on trade-off (substitute) rates ($\Delta z_i/\Delta y$) is necessary. In other words, the decision will be made using the additive model $w_y y + w_{zi} z_i$, where w_y and w_{zi} are the weights for y and z_i, respectively. (See Hayashi et al. (in prep.) for a detailed explanation of the additive model.)

3. CASE STUDIES

3.1 A field experiment on fertilizer rates

This type of studies can be considered as a design problem and thus the purpose of the analysis is to determine the appropriate level of fertilizer applications. We analyze a typical example of wheat production presented by Brentrup et al. (2004). The foreground data are based on the Broadbalk field experiment at Rothamsted with GAP and the background data on ecoinvent 1.3 and IDEMAT2001. The CML 2 baseline method is used as the life cycle impact assessment method. As shown in Figure 1, the result illustrates that the shapes of the functions and models, particularly the ratio model, are different among impact categories. That is, we have multiple optimum points determined by the ratio model.

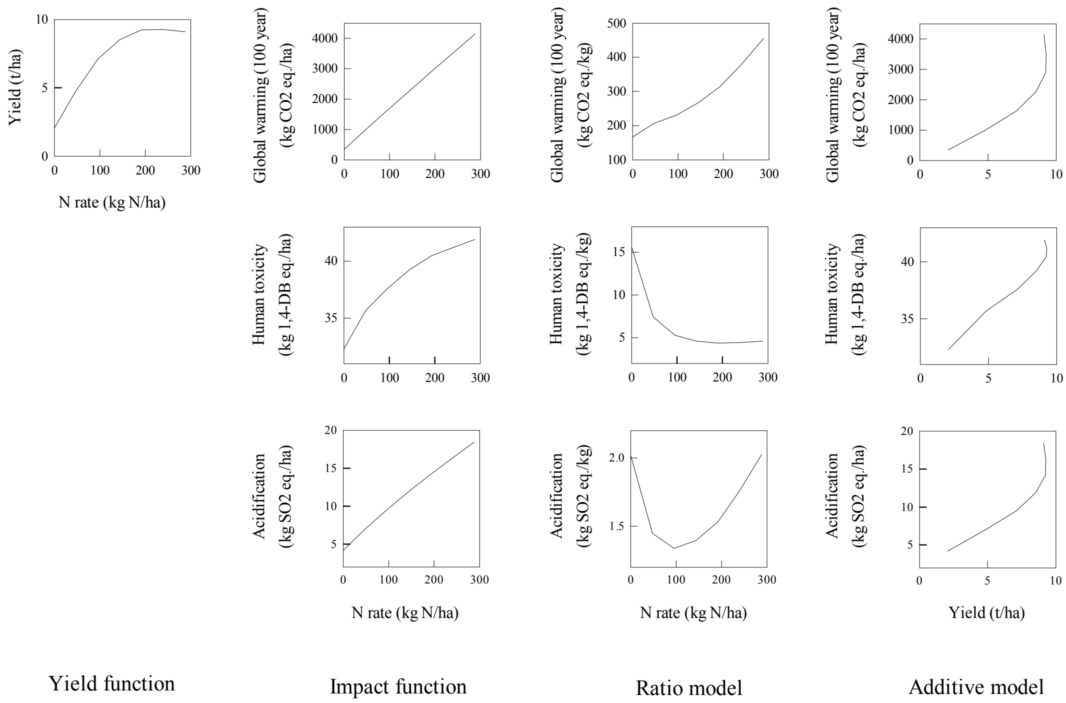


Figure 1. Yield and impact functions and ratio and additive models.

3.2 A field experiment with several agricultural production systems

In this type of the problems, several decision alternatives are defined. Although the alternatives are defined discretely, each alternative can be treated analogous to the above example. The data were adapted from Nemecek et al. (2005) in which the following four alternatives are included: conventional, intensive integrated, extensive integrated, and organic production. The LCIA methods used in this analysis include ecoinvent (for energy), IPCC (for GWP), CML (for human toxicology), and EDIP (for all other categories). The result shows that the impact categories are classified into three groups: (1) global warming and photochemical oxidation, (2) eutrophication and acidification, and (3) toxicity. In the second group, the slopes of the impact functions are negative.

3.3 Pesticide impact assessment using cross-sectional data

The analytical framework presented is already applicable to cross-sectional assessment. We analyzed the farm records of rice farmers in Japan. IMPACT 2002 adapted to Japan was used for calculating the environmental impact (DALYs) of pesticide application. One of the important results is related to the shape of the additive model: if we use the monetary values of the yields, there is no trade-off relationship between the environmental impact per area and yield (gross revenue per area).

4. CONCLUDING REMARKS

Although both the ratio and additive models can be used complementarily for analyzing environmental and economic performance, there are differences between them. An importance difference lies in the possibility of further development. Although the additive model necessitates using preference information (trade-offs between environmental and economic performances) in finding a solution, it can be extended to multi-objective programming models for analyzing the trade-offs.

5. REFERENCES

Brentrup, F., Küsters, J., Lammel, J., Barraclough, P., Kuhlmann, H. 2004. Environmental impact assessment of agricultural production systems using the life cycle assessment (LCA) methodology II. The application to N fertilizer use in winter wheat production systems. European Journal of Agronomy. 20, 265–279.

Cassman, K.G., Dobermann, A. Walters, D.T.,Yang, H. 2003. Meeting cereal demand while protecting natural resources and improving environmental quality. Annual Review of Environment and Resources. 28, 315–358.

Charles, R., Jolliet, O., Gaillard, G.., Pellet, D. 2006. Environmental analysis of intensity level in wheat crop production using life cycle assessment. Agriculture, Ecosystems and Environment. 113, 216–225.

Hayashi, K., Gaillard, G., Nemecek, T., 2006. Life cycle assessment of agricultural production systems: current issues and future perspectives. In: Proceedings of International Seminar on Technology Development for Good Agricultural Practice in Asia and Oceania. National Agricultural Research Center, Tsukuba, Japan, pp. 154–171.

Hayashi, K., Nemecek, T., Scholz, R. in prep. Management intensity, crop yield, and environmental impacts: Integration of economic and environmental performance.

Huppes, G., Ishikawa, M. 2005. Eco-efficiency and its terminology. Journal of Industrial Ecology. 9, 43–46.

Nemecek, T., Huguenin-Elie, O., Dubois, D., Gaillard, G., 2005. Ökobilanzierung von Anbausystemen im schweizerischen Acker- und Futterbau. Agroscope FAL Reckenholz, Zürich; Schriftenreihe der FAL. 58, 155 pp.

Schaltegger, S., Burritt, R. 2000. Contemporary Environmental Accounting: Issues, Concepts and Practice. Greenleaf Publishing, Sheffield, UK.

A dynamic simulation tool for a productive and environmentally efficient food production.

Dr. Johanna Berlin[a], Dr. Karin Östergren[a], Dr. Björn Johansson[b], Asc. Prof. Johan Stahre[b], Prof. Anne-Marie Tillman[c] and M.Sc.Barbro Sundström[a]

[a] SIK-The Swedish Institute for Food and Biotechnology, Department of Environment and Process Engineering, Box 5401, 402 29 Göteborg, Sweden
[b]Chalmers University of Technology, Department of Production Systems, SE-412 96 Göteborg, Sweden.
[c]Chalmers University of Technology, Department of Environmental Systems Analysis, SE-412 96 Göteborg, Sweden.

Abstract
One of the most efficient actions taken by the food industry to reduce food products´ environmental impact is to minimise losses of raw material during production. The reason is that the first step in the lifecycle, the agriculture, is the step that commonly contributes most to the product's environmental impact in a life cycle context.

At the same time new production management concepts are needed in the food industry to increase the diversity of products and lower the costs. A solution is to produce on demand (pull flow). A pull flow often leads to an increased switching of product, which increase the losses.

This contradiction shows that to produce in a cost and environmentally efficient way both aspects must be considered simultaneously. A tool for improving production efficiency based on discrete event simulation and environmental system analysis has been developed. The tool has been used for investigating how environmental impact, cost and revenues are linked to lead-time, production planning, inventory and delivery accuracy in selected food production lines. The tool has been developed on a case study basis and general findings from selected case studies will be presented.

Introduction
The industry is under high pressure to produce a higher variety of products with short lead time at the same time as the environmental concern in society is growing with an increasing speed. Therefore, the industries would in the future have to consider both economy and environment at the same time to be able to stay in market.

Continuous improvements aiming at increased production efficiency is necessary in order to stay in business. By producing on demand from the customer (pull flow) when applicable, inventories can bee minimized and costs reduced. A pull flow will in most cases lead to an increased frequency in production of specific products, which in turn requires flexible and stable production lines. For the food industry these new production management concepts are needed due to a range of factors, including an increasing diversity of products, lower inventories, demands on short lead times and high delivery accuracy.

Environmentally efficient production on the other hand is a matter of societal responsibility which in many cases at the same time leads to a better process economy, e.g. by reducing the loss of raw material during production. For the food industry in particular, loss of raw material significantly raise the environmental impact in a life cycle perspective, due to the large environmental impact of the first step in the lifecycle, the agricultural step. For example, in the life cycle assessment of semi-hard cheese by Berlin (2002) shows that on a life cycle basis the agriculture phase contributed 94% to global warming, 99% to acidification and 99% to eutrophication.

In food production most losses of raw material occur when switching between products and at start up and shut down of a production line, which in many cases favours a decreased production frequency, which is contradictory to the strive for an increased production frequency which has been found to be beneficial of from other aspects. This clearly shows that in order to produce efficiently and environmentally friendly the production efficiency and environmental impact has to be considered simultaneously. Today, the issues of production and environment have been assessed one by one but rarely at the same time.

By combining two existing and widely used approaches for assessing production efficiency; Direct Event Simulation (DES) and environmental impact; Life Cycle Assessment (LCA), it would be possible to enable simultaneous analysis of techno-economic production aspects with environmental consequences of different production options.

In this study we present a general methodology for evaluating production efficiency and environmental efficiency. During development of the methodology we have used a case study approach using DES and LCA as starting points, on three food productions. General findings from the case studies will be presented as well.

Research context

Two areas of research have been combined in the method presented in this paper; LCA and DES to be able to assess industrial production from both cost and environmental perspective in the same simulation model. The combination of DES and LCA is completely new and novel. DES is normally used to gain an overall understanding of complex logical problems, which has dynamics involved. The tool can be used to analyse dynamics in industrial systems such as production, logistics, queuing, processes and resetting. The inputs are cycle times, resource information and random numbers for breakdowns of machines. Output parameters can be utilization of resources, products produced per time unit, buffer size requirements, and other measures needed to dynamically assess cause and effect within manufacturing system.

LCA is a tool for evaluating the environmental impact associated with a product during its life cycle. This is accomplished by identifying and quantitatively describing it's requirements for energy and materials, and the emissions and waste released to the environment. The scope of the assessment is the life cycle, which means that the product under study is followed from the initial extraction and processing of raw materials through manufacturing, distribution, and use, to final disposal, including the transports involved. LCA is an ISO standardised tool (ISO, 2006a and 2006b). More information can be found in Baumann and Tillman (2004) and Berlin (2003).

Method

The method chooses for the methodology development is a systematic combining case research as described in Dubois and Gadde (2002). Combination of theory from LCA and DES was used in reality to various degrees throughout three case studies. A general approach was taken from the start since the methodology was going to be applicable to any food production site. A stepwise approach for the methodology development took place throughout the case studies. The work within and between the steps was made iterative for refining the methodology.

Three different kinds of production, all within the food sector, were selected as case studies; sausage production, fruit juice production and cultured dairy products production. The sausage production was a typical batch production. The fruit juice and the cultured dairy products production can be categorised as semi-continuous productions. The reason of choosing different kinds of products and productions was to get the methodology as general as possible. By choosing production within the food sector we also got the parameters of freshness and shelf life to consider. In each case study a model of the production with its environmental consequences in a life cycle perspective was conducted.

Methodological results

The developed methodology has been made with the purpose to be used to find improvements options in the manufacturing performance. It means that it is changes in the production that will be studied rather than product development or product changes. These improvements options will be identified by simulating changes in the manufacturing performance and its contribution to the environmental impact in a life cycle context, in the very same simulation model. Although the actual changes will take place in the manufacturing step, other parts of the life cycle will be affected as well. Thus, it is crucial to include a life cycle perspective in the environmental calculation. But, as the product will not change, the parts of the life cycle, which comes after the manufacturing process, will be the same in all evaluated improvement options. Therefore, the system boundary in this methodology will include the life cycle from cradle to gate i.e. from raw material acquisition to the product leaving the manufacturing industry, see figure 1.

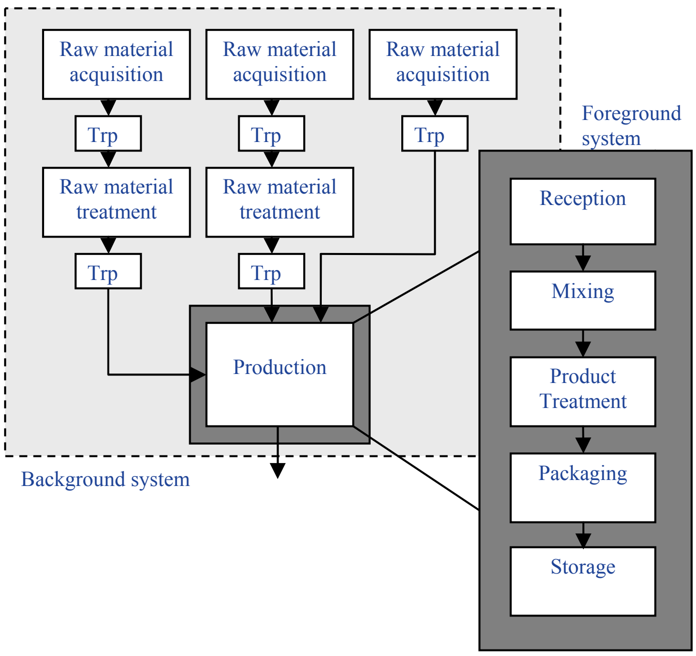


Figure 1. Illustration of the system boundary divided in foreground and background system.

The stepwise working procedure was conducted during the work with the models simulating the production with environmental and production parameters. The working procedure developed covers identification of actors and pin pointing of the exact problems to be solved, knowledge and data search, modelling and verification and finally analysis and conclusions.

Case study result
A number of results was achieved from the simulations for example; the importance of food waste in relation to

- Environmental impact (global warming, acidification, eutrophication and primary energy)
- Production scheduling
- Batch size
- Inventories and
- Delivery accuracy.

An overhaul result from the sausage production is that the spillage of meat has the highest environmental impact followed by energy consumption and packaging. This gives the conclusion that if the sausage industry is going to reduce their environmental impact it is a decrease in spillage which gives the most result. Another general finding which is possible to make is the spillage relation to batch size. It is achievable to test different batch sizes and find out the resulting spillage and its environmental impact independent of production type.

Conclusion
To be able to assess a production from the environmental and economical point of view at the same time in the very same simulation model increase the ability of the stakeholders to make decisions in a more sustainable way.

References

Baumann, H., and A.-M. Tillman. 2004. The Hitch Hiker´s Guide to LCA. An orientation in life cycle assessment methodology and application. Studentlitteratur, Lund , Sweden.

Berlin, J. 2002. Environmental life cycle assessment (LCA) of Swedish semi-hard cheese. International Dairy Journal 12: 939-953.

Berlin, J. 2003. Life cycle assessment (LCA): an introduction. In: Environmentally friendly food processing edited by Mattsson B. and Sonesson U. Woodhead publishing limited, Cambridge, UK.

Dubois, A. and L.-E. Gadde. 2002. Systematic combining: an abductive approach to case research. Journal of Business Research 55 (2002) 553-560.

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

ISO 2006b. Environmental management – Life cycle assessment – Requirements and guidelines. ISO 14044:2006(E). International Organization for Standardization. Geneva. Switzerland

A life cycle approach for improving sustainability in the Norwegian fishing fleet

Ingrid Bouwer Utne

Department of Production and Quality Engineering, Norwegian University of Science and Technology, NO-7491 Trondheim, Norway.
E-mail: Ingrid.b.utne@ntnu.no

Abstract
This paper discusses whether Eco Quality Function Deployment (Eco QFD) can be applied to improve sustainability in the Norwegian fishing fleet. Systems engineering (SE) has been introduced as a feasible process for handling sustainability issues in the fisheries. SE contains methods for general system design, operation, and support in a life-cycle perspective. QFD is related to SE, because it is a method for translating stakeholder needs into detailed system requirements at each life-cycle stage. Life cycle cost (LCC) estimates all future costs related to the system life-cycle. Life cycle assessment (LCA) is a decision support tool supplying information on the environmental impacts of products and processes throughout a product's life cycle. Eco QFD combines QFD, LCC and LCA to evaluate environmental effects and costs in the product development process.

1. INTRODUCTION

Sustainable resource management in an objective of the Norwegian government [1]. Nevertheless, there are several problems, for example, high energy consumption, low profitability, and overcapacity in parts of the fishing fleet; problems indicating that the fish resources are not managed in accordance with sustainable development. In 2004 The Office of the Auditor General concluded that the Norwegian fish resources were not managed consistently with parliamentary resolution and international agreements. The fisheries management's method of work was insufficient because there were no thorough evaluations of the resource control, of the consequences of the resource allocation, the regulations, and the capacity reduction efforts [2]. Moody Marine Ltd. did not recommend Marine Stewardship Council certification of the Norwegian coastal cod and haddock fisheries mainly based on the serious condition of the fish stock [3].

In a systems engineering (SE) perspective, some of the problems for fisheries management are related to identification of stakeholders' needs, and designing a fishery system that fulfils those needs, because the success of a system configuration is related to how well the requirements are specified. SE contains methods for general system design, operation, and support in a life-cycle perspective, and is a combination of technology management and engineering used to control the design of complex man-made systems. In addition to the technical and management elements, SE consists of economic, social, environmental, and political aspects. The SE process is based on identification of the stakeholders' needs and requirements to the system, specification of the system performances and trade-offs taken, before designing and solving [4]. The most difficult and demanding part of the SE process is the definition of the problem and identification of the needs. Quality function deployment (QFD) is related to SE in terms of facilitating communication between the stakeholders and the producer when identifying and ranking the design goals in the SE process [5].

QFD is used to structure product planning and development, and enables the development team to assess the proposed system systematically in terms of how it meets the needs and requirements. The process of QFD involves construction of one or more interlinked matrices, of which the first is called “House of Quality” (HoQ). The HoQ displays the “Voice of the Customer” (VoC) along the left (“whats”), the development team’s technical solutions to the customers’ needs along the top (“hows”), and the impact of the solutions on the needs (relationships) in the middle [6], as shown in Figure 1. The customer may include other stakeholders than only the user [7].

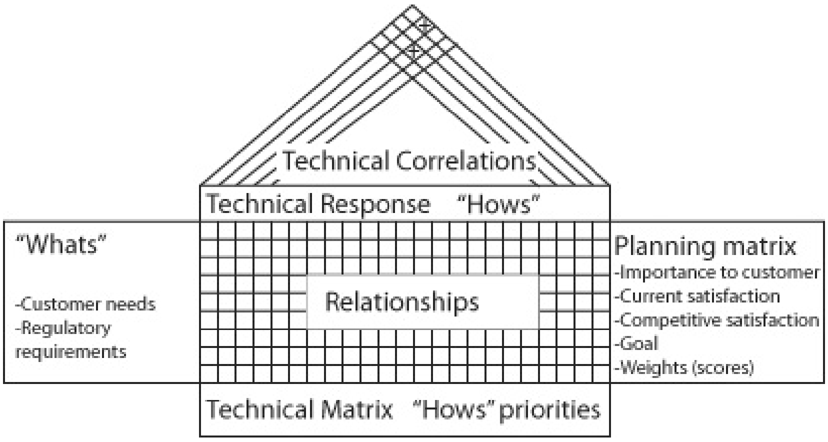


Figure 1. House of Quality

Besides classical QFD, there exist extended approaches which combine environmental issues with customer requirements, called Eco QFD. Examples are green quality function deployment (I, II and III) [8], [9], the quality function deployment for environment [7], and EI2QFD [10]. GQFD II is a further development of GQFD, and integrates life cycle assessment (LCA), life cycle cost (LCC), and QFD into one tool which deploys customer, environmental and cost requirements during the whole development process. The best product alternative can be chosen based on assessing these requirements [8][1]. GQFD III reduces the workload in GQFD II by use of Eco-indicator 99 methodology[2] instead of full LCA, and integrates the analytic hierarchy process (AHP)[3] for selecting the best product/system concept [9].

The objective of this article is to discuss feasibility of Eco QFD for fisheries management and shipyards in the process from need identification to specification of requirements, in order to improve sustainability in the Norwegian fishing fleet.

2. ECO QFD APPLIED TO THE FISHING FLEET- DISCUSSIONS AND CONCLUSIONS

The applicability of using Eco QFD depends on the decision situation.

2.1 Fisheries management

To define requirements for sustainable fisheries is a difficult task, because the concept of sustainability is ambiguous. Fisheries management could use Eco QFD to determine “what is needed” to increase sustainability and to clarify what sustainability in the fisheries is. The fisheries involve many stakeholders, and one of the advantages of QFD is the strong emphasis on stakeholder needs or the VoC. These needs are rated by use of importance scores perceived by the stakeholders.

---

[1] For more information about LCA, see, e.g.: www.pre.nl. About LCC, see, e.g., Fabrycky, W. J. and Blanchard, B. S. (1991). *Life cycle cost and economic analysis*. Prentice Hall, N.J.
[2] For more information about Eco indicators, see www.pre.nl
[3] See, e.g.: Saaty, T. and Kearns, K. P. (1985). *Analytical planning: The organizations of systems*. Pergamon press, Oxford.

For fishers, important needs to a sustainable fishing fleet are, for example, low accident risk and profitability, but these needs may be contradictory or difficult to meet without doing trade-offs.

QFD may function as a way of structuring the needs of the stakeholders and determining which needs should be prioritized. However, determining the "hows" may be more difficult because sustainability in the fishing fleet is not only dependent on the technical characteristics of a fishing vessel, but organizational and regulatory issues as well. Large fishing vessels, such as trawlers have higher fuel consumption than the small conventional vessels, but the trawlers also have a greater distance to their fishing grounds [11]. In Green QFD III, the product options developed in the HoQ are evaluated by the green house matrix for the environmental impacts and the cost house matrix for the life cycle costs. Eco-indicators may be a good idea for environmental impact evaluations of simple products, but regarding fishing vessels, LCA is more applicable, but also more time-consuming.

2.2 Fishing vessel design

Consideration of the costs related to implementation of specific efforts in the fisheries to increase sustainability, for example NOx reducing efforts, could be carried out by use of a cost house as in Green QFD III. However, use of traditional cost/benefit calculations may be more feasible, especially in cases where the product options are already existing solutions, such as in [12]. Fishing vessel owners and shipyards may use Eco QFD to improve the design of fishing vessels, and fishing technology manufacturers may structure their design process when making new products or modifying existing ones. Defining requirements to increase sustainability of fishing vessels means that not only technical and economic needs should be assessed, but environmental issues as well, during the whole life cycle of the vessel.

Nevertheless, classical QFD is time consuming and Eco QFD even more. It may be feasible in development of new products and systems, but challenges occur when applied to complex systems [13]. Eco QFD may increase the environmental quality of products and systems, but the effort for carrying them out often exceeds the limited resources of development projects, even though Green QFD III and EI2QFD are attempts to reduce the demanding work effort.

2.3 Conclusions

For fisheries management it is important to structure and visualize the trade-offs in the decision-making process. Still, when the wanted output is not a pure physical system, for example a "sustainable fishing fleet", Eco QFD seems to be difficult to apply and needs to be adapted. QFD may be used by ship owners, shipyards and manufacturers when designing and constructing fishing vessels and equipment, but to reduce the work load, the HoQ should be limited so that the stakeholders involved focus mostly on controversial issues and the most important decisions.

3. REFERENCES

1. The Norwegian Ministry of Foreign Affairs, *National strategy on sustainable development* 2002.
2. The Office of the Auditor General, *The Norwegian Office of the Auditor General's investigation of the management of the fish resources* 2004.
3. Hough, A., *Pre Assessment Report for Norwegian Whitefish Fisheries*. 2004, Moody Marine Ltd.
4. Utne, I.B., *Systems engineering principles in fisheries management*. Marine Policy, 2006. **30**: p. 624-634.

5. Blanchard, B.S., Fabrycky, W. J., *Systems engineering and analysis*. 1998: Prentice Hall, New Jersey.
6. Cohen, L., *Quality Function Deployment. How to make QFD work for you*. 1995, MA, U.S.A.: Addison-Wesley Publishing Company.
7. Masui, K.S., T., Kobayashi, M., Inaba, A., *Applying Quality Function Deployment to environmentally conscious design.* International Journal of Quality & Reliability Management, 2003. **20**(1): p. 90-106.
8. Zhang, Y., Wang, H. P., Zhang, C., *Green QFD-II: a life cycle approach for environmentally conscious manufacturing by integrating LCA and LCC into QFD matrices.* International Journal of Production Resources, 1999. **37**(5): p. 1075-1091.
9. Mehta, C., Wang, B., *Green Quality Function Deployment III: A methodology for developing environmentally conscious products.* Design Manufacturing, 2001. **4**(1): p. 1-16.
10. Ernzer, M., Mattheir, C., Birkhofer, H. *EI2QFD-an Integrated QFD Approach or From the Results of Eco-indicator 99 to Quality Function Deployment*. in *EcoDesign 2003: Third International Symposium on Environmentall Conscious Design and Inverse Manufacturing*. 2003. Tokyo, Japan.
11. Utne, I.B., *System evaluation of sustainability in the Norwegian cod-fisheries.* Marine Policy, in press, 2007.
12. Norwegian Pollution Authority, *Effort analysis of NOx. Investigation of possible NOx reducing efforts within the energy constructions offshore, in domestic shipping, and in the continental industries* 2006.
13. Hari, A., Kasser, J. E., Weiss, M. P., *How lessons learned from using QFD led to the evolution of a process for creating quality requirements for complex systems.* Systems Engineering, 2007. **10**(1): p. 45-63.

Multifunctional Agriculture: Comparison of food crops and energy crops by Life Cycle Assessment

Kägi T. [a]; Freiermuth Knuchel R. [a]; Gaillard G. [a]; Müller-Sämann K. [b]; Deimling S. [c]; Hölscher Th. [b]

[a] Agroscope Reckenholz-Tänikon Research Station ART, CH-8046 Zurich, Switzerland, gerard.gaillard@art.admin.ch
[b] Agentur für Nachhaltige Nutzung von Agrarlandschaften, D-79379 Mullheim, Germany
[c] PE International, D-70771 Leinfelden-Echterdingen, Germany

Abstract

Conventional agriculture in southern Germany is prone to pollute the groundwater with nitrate. By means of an LCA two options are analysed to reduce the environmental impact on groundwater: i) The introduction of annual energy crops in crop rotations. ii) The production of perennial energy crops. A representative reference crop rotation for food production is compared to a rotation including bioenergy crops (triticale, corn, rape seed) as well as to perennial energy crops namely willow short rotation coppice, miscanthus and meadow. System boundary was set at the farm gate and the respective functional unit was hectare for the function of land cultivation and kg organic dry matter of harvested products for the function of energy production respectively. Results obtained from LCA indicate that the environmental performance of the energy crop rotation, particularly the eutrophication potential and environmental toxicity, is for both functional units inferior to that of the crop rotation for food production. Yet perennial crops such as willow, miscanthus and meadow in general show a distinguishable environmental advantage over crop rotations. The results are leading to recommendations for actions towards a sustainable management of agricultural land for both food and energy production in the Upper Rhine Valley of Germany.

1. Introduction

Regarding the climate change and shortage of fossil energy resources, bioenergy produced from agricultural biomass is more and more supported by governments. Recently, the EU decided to raise the share of renewable energy to 20% of the total energy use in the year 2020. Earlier studies showed that energy production from agricultural biomass was not profitable in Germany [1, 2 & 3]. Nevertheless, energy crops could become competitive not only by considering tax reduction but also by additional incomes like possible contributions linked to ground water protection. In this case, energy crops are likely to replace conventional crops in some areas.

Conventional agriculture in southern Germany is known to pollute the groundwater with nitrate. Our goal is to analyse whether the introduction of annual energy crops in crop rotations or the production of perennial energy crops are, besides reducing fossil energy demand and global warming potential, adequate options compared to conventional food crops to reduce the risk of nitrate leaching and other environmental impacts.

2. Material and Method

The comparison of farming systems at the level of single crops may yield misleading results [4]. The system boundaries are therefore extended to the whole crop rotation. We analysed the following crop production systems:

- **Reference crop rotation (5 years):** winter wheat followed by turnip rape, corn, corn, summer barley, rape seed followed by spontaneous greening
- **Energy crop rotation (5 years):** triticale followed by turnip rape, silage corn followed by mustard, silage corn followed by mustard, summer barley, rapeseed followed by spontaneous greening.
- **Perennial crops (20 years each):** miscanthus; willow (short rotation coppice); permanent meadow

The crop rotations are balanced over 5 years and correspond to typical production scenarios for the studied region. The life span of the perennial crops is 20 years and the results are converted to 5 years for comparison.
The system boundary includes the production of all inputs (mineral fertilisers, machines, pesticides) and all activities on the fields from cultivation to harvest and transport to the farm. Application of farmyard manure was not included since most of the farms in the region do not have animal husbandry. The analysed systems differ in the type and quantity of fertilisers and pesticides applied, in the number of field operations and in yield (Table 1). The production inventories were developed specifically for the project by the local partners.

Table 1: Key characteristics of five cropping systems analysed: Reference and energy crop rotation (CR), miscanthus, willow and permanent meadow (mean values per 5 years).

	CR reference	CR energy	Miscanthus	Willow	Permanent Meadow
Average dry matter yield (kg)	34040	60530	78000	54500	40500
Mineral N-Fertiliser (kg N)	665	468	213	94	150
Fermented biogas substrate (m3)	0	93	0	0	81
Pesticides (kg)	7.6	8.0	0.5	0.6	0.0
N° of Pesticide applications	9.0	8.6	0.3	0.5	0.0
N° of other Field operations	47.5	54.6	15.3	8.4	96.3

For the impact assessment the Swiss Agricultural Life Cycle Assessment tool (SALCA) was used. It includes the environmental inventories of agricultural inputs, taken from [4 & 5], methods developed by the Agroscope Reckenholz-Tänikon Research Station (ART) for the estimation of direct field emissions (more details: http://www.art.admin.ch/themen/00617/00785/index.html?lang=en) and impact assessment methods listed in [4] (see Table2).
Agriculture has to fulfil multiple functions, which cannot be covered with a single functional unit. Hence the following functional units are used: 1) Area (ha), representing the function of land cultivation (sustaining production, landscape, basic life resources) and showing the level of production intensity. As to nitrate leaching this functional unit is our main reference. 2) Organic dry matter yield (oDM), representing the productive function of agriculture from the producer's point of view.

3. Results

The impact assessment shows major differences between the crop rotations and the perennial crops. An overview of the results is given in Table 2.

Table 2: Environmental impacts of the energy crop rotation (CR) and the perennial crops in percent of the reference crop rotation per hectare times 5 years (ha*5a). [dark green = very favourable, light green = favourable, orange = unfavourable, red = very unfavourable; Reading example: The energy demand of miscanthus equals only 49% of the food crop rotation and is therefore very favourable].

Impact category with reference	unit/(ha*5a)	CR reference	CR energy	Miscanthus	Willow	Permanent meadow
Energy demand [6]	MJ-eq	103757	107%	49%	27%	42%
Global warming pot. [7]	kg CO2-eq	18080	95%	36%	19%	32%
Ozone formation [8]	kg Ethylene-eq	3.57	125%	53%	36%	52%
Acidification [8]	kg SO2-eq	92.8	464%	48%	20%	504%
Eutrophication [8]	kg N-eq	344	158%	44%	29%	72%
Terrestrial ecotox. [8]	Tox. points	1809	79%	49%	26%	8%
Aquatic ecotox. [8]	Tox. points	13834	96%	9%	7%	13%
Human toxicity [9]	Tox. points	288	102%	48%	29%	46%
Direct nitrate leaching	kg N	229	122%	41%	26%	0%

Compared to crop rotations perennial crops have lower energy demand, global warming potential and ozone formation due to fewer field operations. Furthermore, they do not need as much mineral fertiliser as crop rotations. The production of mineral fertiliser needs a considerable quantity of non-renewable energy [10] and generates emissions like carbon dioxide, ammonia and nitrous oxide. Miscanthus and willow show better results in acidification and eutrophication due to lower fertilisation and permanent soil coverage that reduces nitrate leaching (see Figure 2). The same is true for the eutrophication of permanent meadow whereas its acidification potential is the highest together with the energy crop rotation. This is primarily due to emissions of ammonia caused by the application of fermented biogas substrate. No pesticide use in the permanent meadow and low pesticide use in willow and miscanthus result in a lower human- and ecotoxicology potential in comparison to that of crop rotations.

Comparing the energy crop rotation with the reference crop rotation there is a higher impact on ozone formation due to more field operations (field cultivation, harvesting and transport of harvest). Intensive field operations combined with less input of mineral fertiliser result in a similar energy use and global warming potential per ha of the two crop rotations (see Figure 1). The acidification and eutrophication potential of the energy crop rotation is higher because of the use of fermented biogas substrate as manure (see Figure 2). In all other impact categories both crop rotations show similar results per hectare. Considering the functional unit kg organic dry matter (oDM) the energy crop rotation has a considerably lower energy demand and a lower eutrophication potential because of the higher yield.

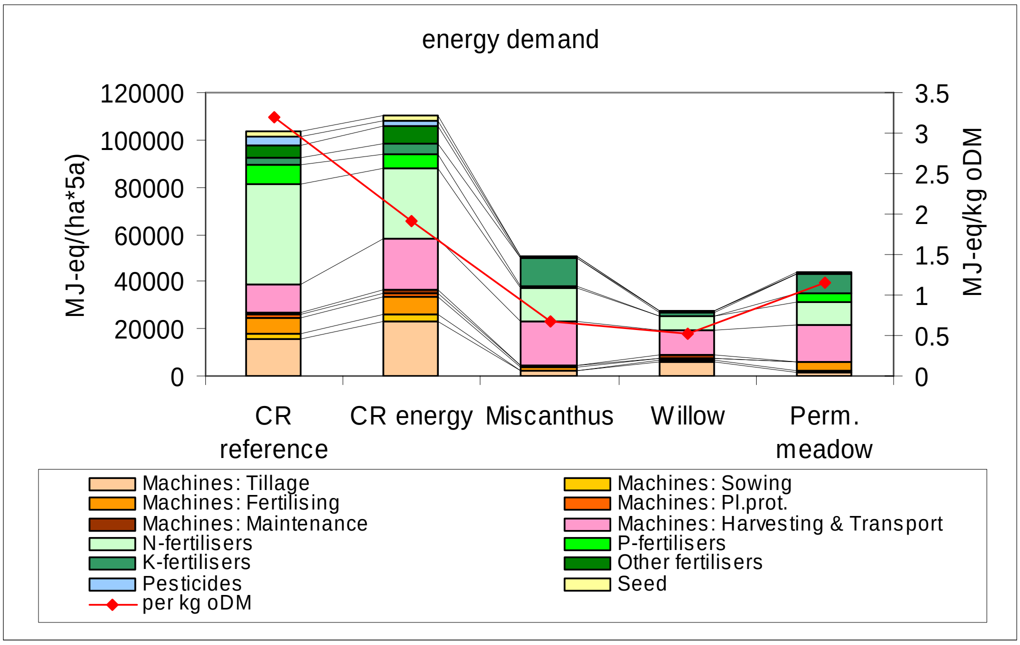


Figure 1: Energy use per hectare times 5 years (ha*5a) and per dry matter unit (oDM)

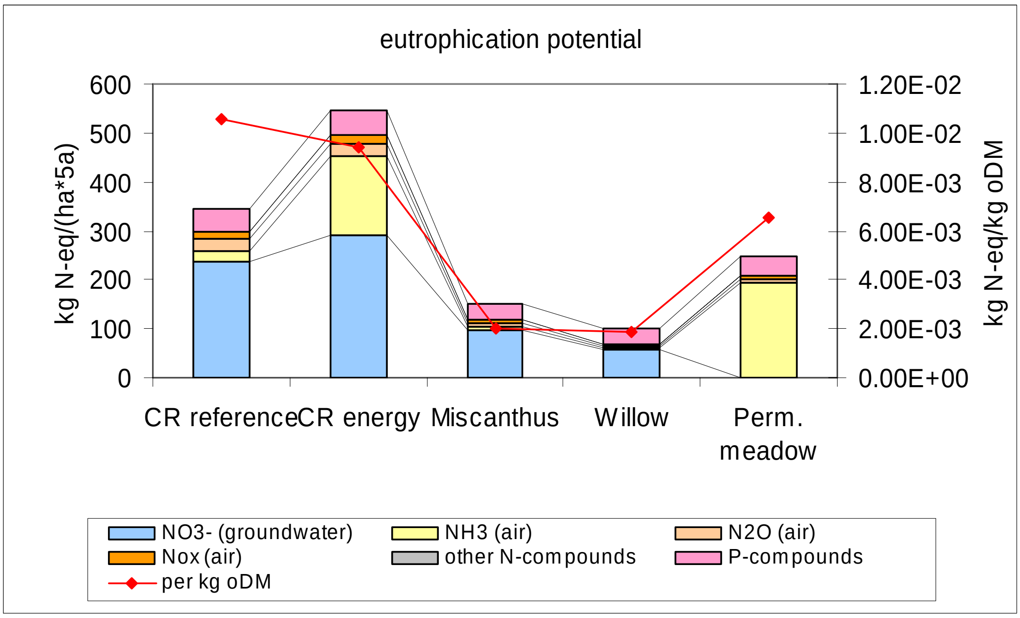


Figure 2: Eutrophication potential per hectare times 5 years (ha*5a) and per unit of organic dry matter (oDM)

4. Conclusions

Regarding ground water protection, the cultivation of perennial crops for energy production is a distinct improvement compared to crop rotations, where intensive soil cultivation and fertilising result in higher potentials of nitrate leaching. On the contrary the analysed energy crop rotation is not suited for ground water protection. It would need further optimisation in the selection and cultivation process of the crops (Extensification of field processes, reducing the amount of fertilisers, crop rotation design). LCA is a suitable instrument to assist in the optimization process.

On arable land the energy crops miscanthus and willow represent an ecologically sound alternative to conventional food production. Permanent meadow for energy production should also be a good option, however the acidification potential is quite high and the yield in oDM remains rather low. Political support of energy crops should therefore mainly focus on perennial energy crops.
The yield of energy crops is higher than the yield of crops for food production, since almost the whole plant is harvested. From a producer's point of view, energy crops are favourable considering the better environmental performances per kg product and by offering interesting opportunities for the diversification of farm products. This study shows that the agricultural function energy crop production can be ecologically competitive to food crop production.

5. Acknowledgements

The project is funded by the innovation fund of badenova AG, Freiburg (Germany).

6. References

1. BMU 1999. Klimaschutz durch Nutzung. erneuerbarer Energien. Studie der DLR/Wuppertal-Institut/ZSW/IWR/Forum für Zukunftsenergien im Auftrag des BMU, Berlin.
2. Öko-Institut [Hrsg.] 2000. Biomasse - Renaissance einer Energiequelle. ökonomy 3/2000
3. Maier, J.; Vetter, R.; Siegle, V. & Spliethoff, H. 1997. Anbau von Energiepflanzen Ganzpflanzengewinnung mit verschiedenen Beerntungsmethoden (ein- und mehrjährige Pflanzenarten); Schwachholzverwertung. Abschlußbericht zum Forschungsvorhaben (Ord.-Nr. 22-94.11), Hrsg. Ministerium Ländlicher Raum, Stuttgart.
4. Nemecek T., Huguenin-Elie O., Dubois D. & Gaillard G. 2005. Ökobilanzierung von Anbausystemen im schweizerischen Acker- und Futterbau. Agroscope FAL Reckenholz, Zürich; Schriftenreihe der FAL 58, 155 p.
5. Frischknecht R., Jungbluth N., Althaus H.-J., Doka G., Hellweg S., Hischier R., Nemecek T., Rebitzer G. & Spielmann M. 2004. Overview and Methodology - ecoinvent data v1.1. Swiss Centre for Life Cycle Inventories (ecoinvent), Dübendorf; ecoinvent report 1, 75 p.
6. Frischknecht R., Jungbluth N., Althaus H.-J., Doka G., Hellweg S., Hischier R., Nemecek T., Margni M. & Spielmann M. 2004. Implementation of life cycle assessment methods - ecoinvent data v1.1. Swiss Centre for Life Cycle Inventories (ecoinvent), Dübendorf; ecoinvent report 3, 116 p.
7. IPCC 2001. Climate Change 2001: The Scientific Basis. In: Houghton, J. T. et al. (eds.), Third Assessment Report of the Intergovernmental Panel on Climate Change (IPCC). IPCC, Intergovernmental Panel on Climate Change, Cambridge University Press, The Edinburgh Building Shaftesbury Road, Cambridge, UK.
8. Hauschild M &. Wenzel H. 1998. Environmental Assessment of Products: Scientific background. Chapman&Hall, London, 565 p.
9. Guinée J. B., Gorrée M., Heijungs R., Huppes G., Kleijn R., de Koning A., van Oers L., Wegener Sleeswijk A., Suh S., Udo de Haes H. A., de Bruijn H., van Duin R., Huijbregts M. A. J., Lindeijer E., Roorda A. A. H. & Weidema B. P. 2001. Life cycle assessment - An operational guide to the ISO standards. Ministry of Housing, Spatial Planning and Environment (VROM) and Centre of Environmental Science (CML), Den Haag and Leiden, Netherlands.
10. Nemecek T. & Erzinger S. 2005. Modelling Representative Life Cycle Inventories for Swiss Arable Crops. Int J LCA, 10: 68-76.

Life cycle inventory of different forms of rice

Poritosh Roy[a], Tsutomu Ijiri[b], Daisuke Nei[a], Hiroshi Okadome[c], Nobutaka Nakamura[a], Yutaka Ishikawa[a] and Takeo Shiina[a,]*

[a]Distribution Engineering Laboratory, National Food Research Institute, National Agriculture and Food Research Organization, Kannondai 2-1-12, Tsukuba-shi, Ibaraki 305-8642, Japan
[b]Forage Rice Research Team (Kanto Region), National Agricultural Research Center, National Agriculture and Food Research Organization, Kannondai 3-1-1, Tsukuba-shi, Ibaraki 305-8666, Japan
[c]Agriculture, Forestry and Fisheries Research Council Secretariat, The Ministry of Agriculture, Forestry and Fisheries, Kasumigaseki 1-2-1, Chiyoda, Tokyo 100-8950, Japan, *shiina@affrc.go.jp

Abstract
Rice is consumed in various forms (milled, partially milled, brown, parboiled milled and germinated brown etc), which consume different amount of resources and energy in the life cycle of rice. This study attempts to evaluate the life cycle of rice (milled, partially milled, brown and parboiled milled) to determine if the environmental load from the life cycle of rice can be reduced. This study revealed that the parboiled rice consumed more energy compared to others, hence emit greater amount of CO_2 followed by brown, milled and partially milled rice. It is worthwhile to note that the partially milled rice emits slightly lesser amount of CO_2 compared to the well-milled, brown and parboiled rice. A change in rice production and consumption patterns would be helpful to reduce environmental pollutions. However, taste of rice and its acceptability need to be considered to adopt the lower degree of milling.

Keywords: rice, consumption patterns, environment and LCA.

1. Introduction

The food industry is one of the world's largest industrial sectors, hence a large user of energy. While food processing is not considered to be amongst the most environmentally hazardous industries, nevertheless, they can cause severe pollution if designed and operated with insufficient attention to the environment (Ramjeawon, 2000). Rice is a staple food for nearly two-thirds of the world's population. In Japan, rice is considered as the most important crop in domestic agriculture. Japan produced 11.34 million tones of rough rice (paddy) in 2005. However, rice consumption has been decreasing since 1960s, leads to develop different value added rice products (germinated brown rice, rice bread etc.) to stabilize the rice consumption. The rice-based foods have also been diversified in the forms of frozen or aseptic cooked rice products. Various types of rice and rice products emit different amount of CO_2 in the life cycle of rice. Agriculture and in particular rice production has been named as a significant contributor to greenhouse gases emission (primary and secondary emission). Primary emission is related directly to rice culture and depend on the metabolism of the rice plant and the soil where it grows. Secondary emission is related to the inputs in rice production, including agricultural machinery, fertilizer, pesticides and others (Breiling, et al., 2005). Therefore, there is a need to evaluate the life cycle of rice and rice products to determine an energy effective rice consumption patterns to reduce CO_2 emission.

Greenhouse gas emission (GHG) has been increasing remarkably by a tremendous use of energy resulted global warming, is perhaps the most serious problem mankind faces today. It is also predicted that global warming will make rice crops less productive (Sheehy et al., 2005) and it could threaten to erase the hard-won productivity gain which have so far kept the rice harvest in step with population growth. Under the United Nations Framework Convention on Climate Change (Kyoto Protocol), countries have been agreed to stabilize the CO_2 emission to the 1990 level, which needs to reduce from its present level. Japan has set a target to reduce her GHG emission 6% below the 1990 level by 2008 to 2012. Reduction of GHG emissions is a challenging task, as the growth of the economy tends to increase the energy demand, in consequences increase the CO_2 emission. Therefore, this study attempts to evaluate the life cycle of rice (milled, partially milled, brown and parboiled milled) to determine if CO_2 emission can be reduced.

2. Methodology

The purpose of an LCA (life cycle assessment) can be: comparison of alternative produces, processes or services; comparison of alternative life cycles for a certain product or service; identification of parts of the life cycle where the greatest improvement can be made. The LCA methodology categorized in four steps: goal definition and scoping, inventory analysis, impact assessment and interpretation (ISO, 1997). However, this study deals only with the first two steps i.e., goal definition and scoping, and the inventory analysis. The goal of this study was to investigate the life cycle of different forms of rice consumed at households to quantify and to evaluate the environmental loads and compare them to facilitate decision making. Figure 1 shows the flow diagram of the life cycles of different forms of rice. The functional unit of this study is defined as the mass of the product i.e., 1 kg of rice (milled or partially milled or brown or parboiled milled) consumed at households.

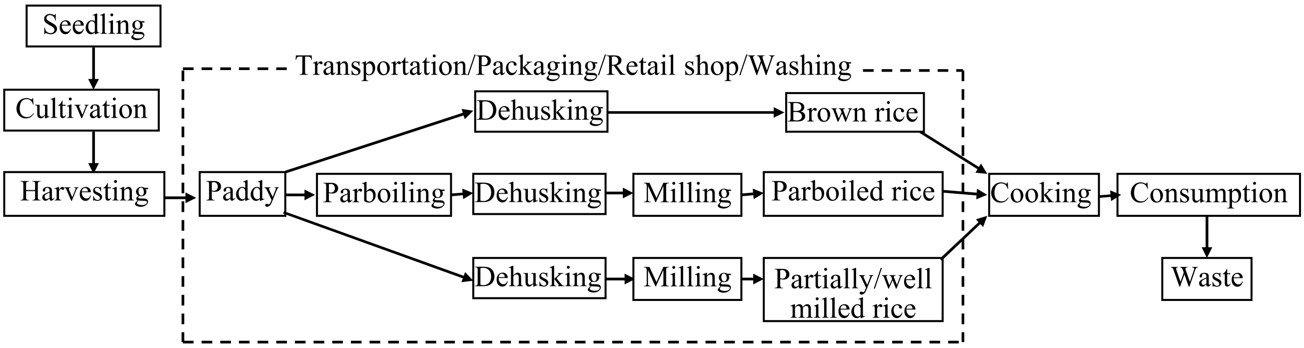


Fig. 1 Flow diagram of the life cycles of different form of rice

Although energy and resources consumption varies on the location of production, production seasons and variety of rice, CO_2 emission/kg of paddy was assumed to be the same for both the Japonica and Indica rice for this study because of the lack of data. For the inventory analysis some of the data were collected from the literature and others were measured. The emission from Japonica rice production (NIAES, 2003), packaging, transportation and waste management (Tahara, 2005) were collected from the literature. The emission from the parboiling process of Indica variety was also collected from the literature (Roy et al., 2005). Energy consumption in the milling and coking processes were measured.

2.1 Measurement of energy consumption in milling and cooking processes

Energy consumption in the milling process of well-milled rice (degree of milling 10%) was measured at a local agricultural farm (Kubota milling machine; Power meter: A35W, Osaka Electric Co., Ltd.), however the energy consumption in the milling process of partially milled rice was measured in the laboratory (Milling machine: Hosokawa, RK30; Power-meter: WT110E digital power meter, Model no. 2534GA251 F, Yokogawa). The cooking processes of milled, partially milled, brown and parboiled milled rice were also studied at the laboratory. Different amount of rice was cooked using rice cookers (Zojirushi, NH-PA10-HH and NP-GA05-XA; Power meter: WT110E, Yokogawa). The rice was washed and soaked for one hour at room temperature (24-25˚C) before being cooked. The water temperature was 25˚C. The rice-water ratio was 1.5, 1.6, 1.875 and 2.5 for milled, partially milled, brown and parboiled milled rice (basmati), respectively. The quality indices (moisture content and hardness) of cooked rice were also measured.

3. Results and Discussion

3.1 Energy consumption in milling and cooking processes

In the life cycle of rice, various types of final energy have been consumed. The mechanical energy is used for milling process and in the case of cooking the thermal energy is used. The energy consumption in the milling process was found to be 0.0233 kWh/kg-brown rice. The energy consumption in the milling process of parboiled rice was reported to be 0.0263 kWh/kg (Roy et al., 2005). The energy consumption in the cooking process was varied on the amount of rice cooked and the capacity of the rice cooker (Fig. 2). The higher the amount of rice cooked the lower energy consumption per functional unit was observed. Therefore, the energy consumption in the cooking processes at the full capacity of the rice cooker (PA-GA05-XA) was used to calculate CO_2 emission. However, the brown rice consumed the highest amount of energy (0.571 kWh/kg) followed by the parboiled (0.433 kWh/kg), partially milled (0.352 to 0.367 kWh/kg) and milled rice (0.333 kWh/kg). The difference in energy consumption might be because of the types of rice, water-rice ratio, and the cooking mode. The moisture content and the hardness of cooked rice varied from about 60 to 65% and 11 to 15 N, respectively, which seems to be dependent on the type of rice and the cooking conditions.

3.2 CO_2 emission from the different stages of rice life cycle

The atmospheric emission is directly related to the energy consumption. Based on the material and energy consumption at the different stages of the life cycle of rice, the CO_2 emission was calculated (Fig. 3). The production stage emits the highest amount of CO_2 among all the stages of the life cycle of rice followed by the cooking and the others in the case of untreated rice. Similarly, in the case of parboiled rice the emission was also found to be the highest for production followed by the parboiling stage and the others. The emission was found to be the highest for the parboiled rice followed by the brown rice and the lowest was the partially milled rice (degree of milling, 2%). The emission was greater for brown rice than that of milled and partially milled rice even no milling is required, because of the higher energy consumption during cooking compared to others due to the greater water-rice ratio and longer cooking time. The environmental load of partially milled rice was found to be slightly lower than that of well-milled rice because of the difference in head rice yield, which leads to lower emission from its production stage. The production stages were found to be the hotspots for both the untreated and parboiled rice.

This study reveals that all the processes have a negative effect on the environment and the intensity of environmental loads dependent on the production and consumption patterns. Substitution of rice production and consumption pattern is required to reduce environmental load. The partially milled rice is found to be environmentally-friendly compared to the others. Milled rice option required greater amount of paddy compared to that of brown rice due to the milling treatment, however partially milled rice option required lesser amount of paddy compared to the well-milled rice because of the lower degree of milling. The partially milled rice not only reduces the environmental load but also increases the head rice yield compared to that of well-milled rice. However, taste of rice and its acceptability need to be considered to adopt lower degree of milling.

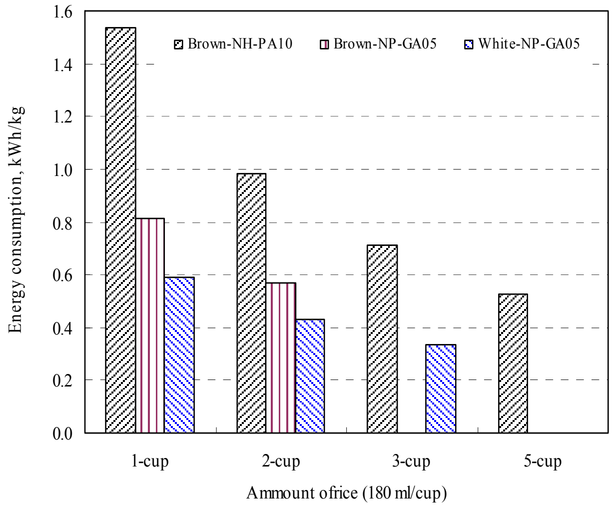


Fig. 2 Energy consumption in the cooking process

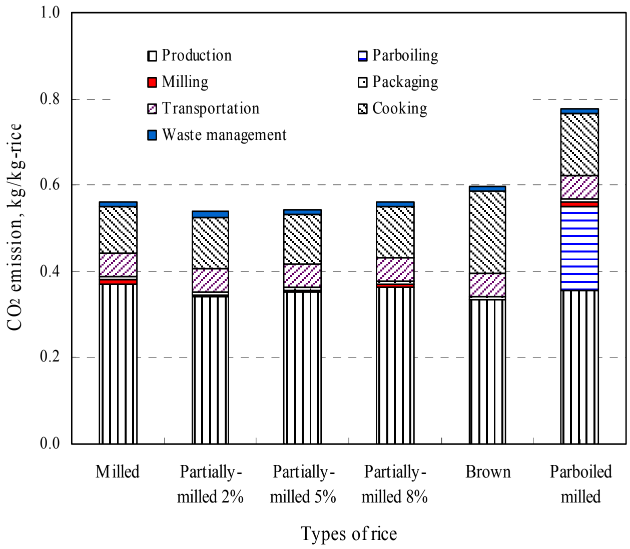


Fig. 3 LCI of different types of rice

4. Conclusions

This study makes it possible to compare the environmental loads of different types of rice and it reveals that the environmental load is dependent on the production and consumption patterns. The partially milled rice (degree of milling 2%) option was found to be the most environmentally-friendly compared to the others. Thus, the substitution of rice production process and consumption pattern would reduce the energy consumption and atmospheric emission. Motivation and awareness on environment and health are required for method switching. However, taste of rice and its acceptability need to be considered to adopt lower degree of milling.

Acknowledgement: This work is supported by the Japan Society for the Promotion of Science.

References

Breiling, M., Hashimoto, S., Sato, Y., and Ahamer, G., 2005. Rice-related greenhouse gases in Japan, variations in scale and time and significance fro the Kyoto Protocol, Paddy Water Environment, 3: 39-46.

ISO (International Organization for Standardization), 1997. ISO 14040 Environmental management-life-cycle assessment-principles and framework.

NIAES (National Institute for Agro-Environmental Sciences), 2003. Report on the research project on life cycle assessment for environmentally sustainable agriculture, Ibaraki, Japan.

Ramjeawon, T., 2000. Cleaner Production in Mauritian Cane-Sugar Factories. Journal of Cleaner Production, 8(6): 503-510.

Roy, P., Shimizu, N. and Kimura, T., 2005. Life cycle inventory analysis of rice produced by local processes, Journal of the Japanese Society of Agricultural Machinery, 67(1): 61-67.

Sheehy, J. E., Elmido, A., Centeno, G. and Pablico, P., 2005. Searching for new plants for climate change, Journal of Agricultural Meteorology, 60(5):

Tahara, K., 2005. Annual report of the food study working group for sustainable consumption, The Society of Non-Traditional Technology, Japan.

Environmental effects and consumer considerations of consuming lettuce in the UK winter

L. Milà i Canals [a]; A. Hospido [b]; R Clift [a]; M Truninger [c]; B Hounsome [d]; G Edwards-Jones [e]

[a] Centre for Environmental Strategy, University of Surrey (D3), Guildford GU2 7XH, UK
[b] Department of Chemical Engineering, University of Santiago de Compostela, 15782, Spain
[c] School of Social Sciences, University of Wales, Bangor LL57 2DG, UK
[d] Centre for Economics and Policy in Health, University of Wales, Bangor LL57 1UT, UK
[e] School of the Environment and Natural Resources, University of Wales, Bangor LL57 2UW, UK

Abstract

Winter consumption of lettuce has increased in the UK, simultaneously with reduced winter domestic production in heated glasshouses, supplied by increased imports from countries (primarily Spain) where lettuce can be produced outdoors over winter. This has fuelled the debate on food miles, in which some groups argue that the distance food travels must be reduced. While some consumers favour local food, most expect access to produce year-round and are not ready to substitute lettuce by local alternatives such as chicory or winter cabbage. This paper compares the environmental consequences of out-of-season domestic lettuce production in UK heated glasshouses with outdoor winter production in Spain imported by road into the UK. Out-of-season production in the UK has higher energy use and related impacts (e.g. global warming potential) than lettuce imported from Spain, whereas the land and water use tends to be lower in protected cropping. More research is needed into the likely effects of potential winter substitutes for lettuce.

1. INTRODUCTION

Research to compare localised and globalised food supply [1-6] has contributed to debate around the 'food miles' concept [1-6], i.e. 'the distance food travels, from the farm to consumer' [1]. Year-round supply of fresh produce reflects supermarkets' sourcing strategies and consumer demands: according to recent research [7], over 70% of consumers in UK urban areas expect to be able to purchase any product at any time of year. This study explores the following questions:

- How do the environmental impacts related to providing 1 kg of lettuce to the British consumer vary through the year?
- What are the environmental consequences of supplying lettuce year-round?
- Are there alternatives associated with lower environmental impacts?

2. MATERIALS

Lettuce consumption in the UK has been assessed considering the main sources through the year, following recommended approaches to scenario definition to compare domestic with imported food [6, 8]. Open field production in the UK is considered for supply from May to October, whereas lettuce is imported, primarily from Spain, during the rest of the year. Table 1 summarises the supply sources considered for the different months of the year. 1st and 2nd crop indicate whether farming practices for early or late crops are considered:

- UK-1: UK 1st crop (farms: a, b, c);
- UK-2: UK 2nd crop (farms: a, b, c);
- UK-In: UK indoor (glasshouse) production (farms: c, d);
- ES-1: Spain 1st crop (farms: a, b);

– ES-2: Spain 2nd crop (farms: a, b)

Outdoor production practices change through the seasons to respond to weather conditions; e.g. UK early crops (harvested May to mid July) are protected with fleece to prevent frost damage, and water consumption is higher in later months, while early Spanish crops (planted in August-September) generally require more water for irrigation. A large retailer perspective has been adopted, because large retailers supply 80% of fresh produce in the UK. Major retailers use Regional Distribution Centres (RDC), so that downstream stages are the same regardless of the origin of the product. Therefore, the scope of the study is cradle-to-RDC; different results could be expected for more localised distribution, such as local or farmers' markets, because of the differences in the distribution networks.

Table 1: Sources of lettuce for UK consumption through the year

Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
UK-In ES-2	UK-In ES-2	UK-In ES-2	UK-In ES-2	UK-1	UK-1	UK-1 UK-2	UK-2	UK-2	UK-2	UK-In ES-1	UK-In ES-1

3. METHODS

The system for LCA extends from plant propagation to the Regional Distribution Centre (RDC), including all farm operations and post-harvest cooling. Lettuce sold loose has been studied, and no packaging production has been considered. The study includes production of cos, iceberg and green oak leaf lettuces and fine endives. The functional unit is 1 kg of lettuce delivered to a UK RDC. No distinction has been made on lettuce variety or nutritional content (actually, only certain varieties can be grown in glasshouses). Different producers grow lettuces to different sizes depending on costumer requirements; this would have an effect on the results, e.g. through yields, but is not considered here.

Data on farm production practices, post-harvest cooling and transport to RDC have been gathered directly from individual producers in the UK (3 for open field: UKa, UKb and UKc; 2 for under-glass: UKc-In and UKd-In) and Spain (2 producers: ESa and ESb). All other data have been obtained from the ecoinvent 1.2 database. Field emissions have been calculated following accepted approaches [9, 10]. Transportation of farm workers has been included for labour-intensive operations in Spain; this information is still not available in the UK, but its contribution to the final results is minimal. Impact assessment used the CML 2001 method [11]. The results for two relevant impact categories (Eutrophication, EP; and Global Warming Potential, GWP) are complemented with results for:

- Primary non-renewable Energy Use, PEU (measured in MJ)
- Land Use, LU (measured in m^2year)
- Water Use, WU (measured in litres)

LU and WU in particular need further characterisation for proper interpretation (e.g. incorporating the effects of different land management practices on soil quality [12] or the source and regional scarcity of water), but they are included here for illustration.

Data on consumers' attitudes to salad were collected from 7 focus groups (40 consumers in total) and over 50 in-depth interviews with residents in three rural areas in the UK: Anglesey, Herefordshire and Lincolnshire [13].

4. RESULTS

Figure 1 shows the relative contribution of each supply scenario to the assessed impacts and indicators through the year, expressed as % of the biggest contribution by any supplier, while Table 2 shows the absolute contributions.

It is noteworthy that the relative contributions to PEU and GWP from indoor production are overwhelmingly larger than from field production, even for imported produce, primarily due to energy for heating and (to a much lesser extent) lighting the glasshouses. Actually, high energy use and rising energy prices explain why many British indoor lettuce growers have discontinued production, often acting as importers during winter. Around 40-50% of PEU and GWP in Spanish lettuce come from transportation to the UK. Transportation of farm workers has a negligible effect (<1%) on the results. Impacts on Eutrophication (EP) show similar contributions regardless of supplier, with variations due to different yields per ha and different use of fertilisers.
Although Spanish winter production and UK summer production are not comparable because they come in different seasons and so deliver different products from a consumer perspective, it is quite striking that energy use is not much larger for Spanish imports than for UK-grown lettuce; they are of comparable magnitude and UKa-1 actually uses more energy than ESb-2. However, it should also be noticed that differences between farms within the same country can be as large as between domestic and imported lettuce.
Land (LU) and water use (WU), are dominated by the agricultural stage, as expected. However, it should be noted that 4-6% of the land used is sealed or otherwise heavily transformed (roads, buildings, etc.), and should arguably be weighted more heavily in the characterised results. Glasshouses are used more intensively than open fields (e.g. UKc-In and UKd-In produced 3 and 5 crops per year, respectively), and therefore the values for LU in indoor production tend to be (2-5 times) lower than for outdoor.

Table 2: Contributions of the studied suppliers to the assessed impact categories and indicators. All values are expressed per 1 kg of lettuce delivered to the RDC.

		Impact Categories		***Inventory Indicators***		
Season	**Supplier**	**GWP 100 years [kg CO2-Eq]**	**EP [kg PO43-Eq]**	**PEU [MJ]**	**LU [m^{2}year]**	**WU [litres]**
Jan-Apr	**UKc-In**	3.72E+00	7.94E-04	7.90E+01	6.66E-02	3.83E+01
	UKd-In	1.18E+00	3.19E-04	2.48E+01	5.70E-02	1.28E+01
	ESa-2	5.41E-01	1.02E-03	6.86E+00	1.53E-01	8.53E+01
	ESb-2	3.56E-01	6.37E-04	5.31E+00	1.47E-01	2.88E+01
May-Jul	**UKa-1**	4.14E-01	8.38E-04	5.85E+00	3.37E-01	3.94E+01
	UKb-1	2.85E-01	4.27E-04	3.90E+00	1.87E-01	2.67E+01
	UKc-1	3.50E-01	7.29E-04	4.32E+00	3.18E-01	5.70E+01
Jul-Oct	**UKa-2**	2.88E-01	5.43E-04	4.42E+00	2.55E-01	3.60E+01
	UKb-2	2.28E-01	3.31E-04	3.52E+00	1.87E-01	4.83E+01
	UKc-2	3.50E-01	7.25E-04	4.10E+00	3.18E-01	8.27E+01
Nov-Dec	**UKc-In**	3.72E+00	7.94E-04	7.90E+01	6.66E-02	3.83E+01
	UKd-In	1.18E+00	3.19E-04	2.48E+01	5.70E-02	1.28E+01
	ESa-1	5.22E-01	1.01E-03	6.43E+00	1.52E-01	5.10E+01
	ESb-1	3.94E-01	8.03E-04	6.40E+00	1.77E-01	1.02E+02

Water use is often higher in outdoor production, particularly in the hottest months: lettuce imported from Spain in November and December has been on the field during a very hot period (August and September), requiring more than twice as much water as British indoor production. Also UK production over summer (2nd crop) requires more water than indoor production, and UKc-2 shows a value for WU twice that of UKc-In.

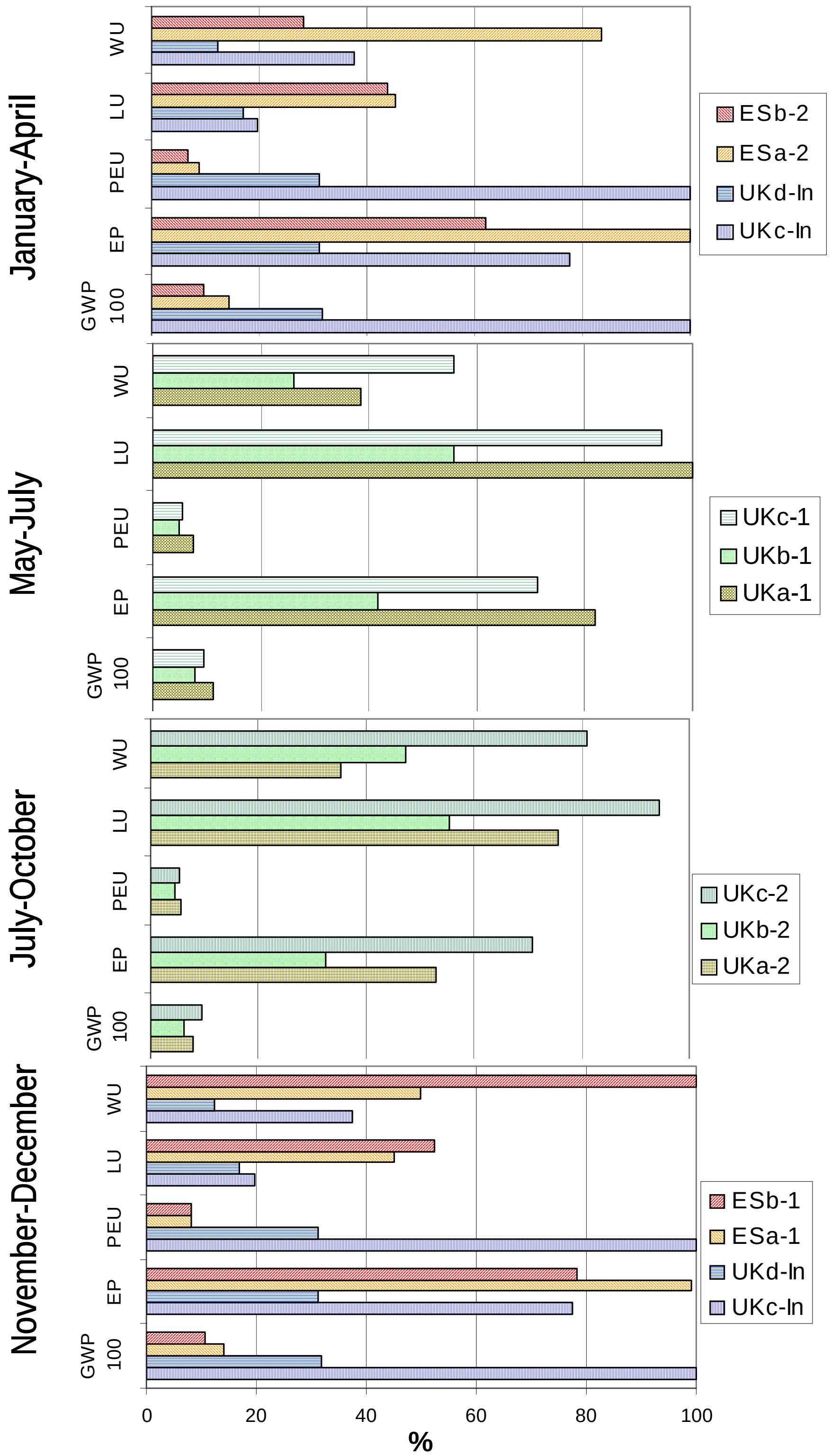


Figure 1: Relative contributions of the studied suppliers to the assessed impact categories and indicators, as % of the largest contribution to each impact/indicator through the year.

In the sample of consumers interviewed for this study, the most striking seasonal variation in vegetable/salad eating was a tendency to consume more salads in the summer and more vegetables in the winter. When presented with a hypothetical choice to replace lettuce in the winter for other vegetables (e.g. cabbage or chicory), consumers found this decision not relevant to their daily lives as a considerable proportion of them tend not to eat lettuce in the winter (apart from special occasions or when eating out). Lettuce and cabbage were generally not considered equivalent products, as they demand different methods of preparation and cooking and have different taste and texture. Coleslaw was the predominant home-prepared winter salad not containing lettuce.

5. DISCUSSION

Many "common" LCA impact categories and PEU show clearly higher contributions for indoors lettuce production over winter in the UK when compared to lettuce imported from Spain, with the exception of Eutrophication. The latter has been modelled in a relatively simple way, allocating fixed values per ha for nitrate and phosphate leaching. When lettuce is imported from Spain during winter, on the other hand, only a slight increase in PEU and no clear difference in GWP or EP are observed when compared with summer outdoor production in the UK. The most significant differences lie in LU and particularly WU, but WU needs further characterisation to capture the significance of water scarcity. Also the specific effects of occupying 1 m2year with arable land or with a glasshouse need to be factored in the assessment.
The sample of consumer interviews does not provide an insight into why winter consumption of lettuce is increasing in the UK. It appears likely that the increase in winter consumption is driven by restaurants and food caterers (e.g. as filler in sandwiches). If this is the case, then the comparisons made in this paper would suggest that continuing the increase in imports from Spain would reduce the contribution to some of the environmental impacts (GWP, PEU) compared to UK indoor production. With the trend towards increased imports an increase in LU and WU would be expected. However, other alternatives such as cabbage or chicory would need to be explored too. Preliminary results suggest that a shift from imported lettuce to chicory might induce higher energy use. Winter cabbage production still needs to be assessed. Furthermore, if a consumer perspective is followed, then the alternative winter salads should be studied in more detail. For example, if coleslaw is considered, then production of mayonnaise and cream (amongst other ingredients) should be assessed for a proper comparison with (imported) lettuce-based salads. It should be highlighted that consumer interviews were all based in rural areas and reveal a preference for local and seasonal food, whereas recent research suggests these results could vary significantly in urban areas [7].

The variability in the results between the different farms studied suggests that there is ample potential for improvement in both indoor and outdoor production, and also for imported produce. These results, in particular the likely environmental hotspots in lettuce production, need to be properly communicated to industry to guide reducing environmental impacts.

6. ACKNOWLEDGEMENTS

Drs Milà i Canals, Truninger and Hounsome are funded by the Rural Economy and Land Use programme (http://www.relu.ac.uk). Dr Hospido is funded by the Isidro Parga Pondal programme (Xunta de Galicia).

7. REFERENCES

[1] Smith A, Watkiss P, Tweddle G, McKinnon A, Browne M, Hunt A, Treleven C, Nash C, Cross S (2005): The Validity of Food Miles as an Indicator of Sustainable Development. ED50254, -103. 2005. Oxon, UK, Defra

[2] Schlich EH, Fleissner U (2005): The Ecology of Scale: Assessment of Regional Energy Turnover and Comparison with Global Food In: Int J LCA **10**(3)219-223

[3] Jungbluth N, Demmeler M (2005): 'The Ecology of Scale: Assessment of Regional Energy Turnover and Comparison with Global Food' by Elmar Schlich and Ulla Fleissner In: Int J LCA **10**(3)168-170

[4] Sim S, Barry M, Clift R, Cowell SJ (in press): The Relative Importance of Transport in Determining an Appropriate Sustainability Strategy for Food Sourcing. A Case Study of Fresh Produce Supply Chains. Int J LCA 2006 (OnlineFirst): 10 pp. DOI: http://dx.doi.org/10.1065/lca2006.07.259
[5] Blanke MM, Burdick B (2005): Food (miles) for Thought. Energy Balance for Locally-grown versus Imported Apple Fruit. Environ Sci and Pollut Res **12**(3): 125-27
[6] Milà i Canals L, Cowell SJ, Sim S, Basson L (submitted): Comparing Domestic versus Imported Apples: A Focus on Energy Use. Environmental Science and Pollution Research
[7] Barry Hounsome, University of Wales, personal communications November 2006
[8] Edwards-Jones G, Cowell SJ, Cross PA, Day G, Hospido A, Hounsome B, Ivashikina N, Jones DL, Koerber G, Milà i Canals L, Tomos D, Truninger M, Tudor-Edwards R, York EH (in preparation): *The ethics of purchasing food – it's more than the mileage*. Trends in Food Science and Technology
[9] 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 [http://www.sri.bbsrc.ac.uk/science/bmag/bmaghome.htm#Lifecycleassessments]
[10] Nemecek T, Heil A, Huguenin O, Meier S, Erzinger S, Blaser S, Dux D, Zimmermann A (2004): Life Cycle Inventories of Agricultural Production Systems. Ecoinvent Report No. 15. Agroscope FAL Reckenholz and FAT Taenikon, Swiss Centre for Life Cycle Inventories, Dübendorf (Switzerland), for ecoinvent members only
[11] Guinée J (ed), Gorrée M, Heijungs R, Huppes G, Kleijn R, de Koning A, van Oers L, Wegener Sleeswijk A, Suh S, Udo de Haes HA, de Bruijn H, van Duin R, Huijbregts MAJ, Lindeijer E, Roorda AAH, van der Ven BL, Weidema BP (2002): Life cycle assessment. An operational guide to the ISO standards. VROM & CML, Leiden University (The Netherlands)
[12] Milà i Canals L, Romanyà, J Cowell SJ (2007): *Method for assessing impacts on life support functions (LSF) related to the use of 'fertile land' in Life Cycle Assessment (LCA).* Journal of Cleaner Production. **15** 1426-1440
[13] Truninger, M. and Day G. (2006): Consumers and Local Food: Analysis of Focus Groups with consumers in Anglesey, Herefordshire and Lincolnshire, report prepared under the RELU project

Environmental impacts of introducing grain legumes into European crop rotations

Thomas Nemecek[a], Julia-Sophie von Richthofen[b], Pierre Casta[c], Gaëtan Dubois[d]

[a]Agroscope Reckenholz-Tänikon Research Station ART, CH-8046 Zurich, Switzerland, thomas.nemecek@art.admin.ch
[b]proPlant GmbH Münster, Germany
[c]ITA Valladolid, Spain
[d]UNIP Paris, France

Abstract

The high dependence of Europe on soya bean imports could be reduced by promoting European grain legume production. A life cycle assessment in four regions was performed to study the environmental impacts of an increased grain legume production.

The introduction of grain legumes in intensive crop rotations with a high proportion of cereals and a high level of N-fertilisation reduced significantly the energy demand, global warming potential, ozone formation and acidification as well as eco- and human toxicity per area unit. Nitrate leaching tends in general to be higher, but can in many cases be reduced by including catch crops or sowing winter grain legumes. Due to the lower yields of grain legumes compared with cereals, the advantages of grain legumes are smaller when considered per GJ gross energy of the harvested products. In low-input crop rotations, no advantage of introducing grain legumes was found.

1. INTRODUCTION

Over 75% of the materials rich in proteins, used for feeding, need to be imported to Europe, mainly in form of soya bean meal of mainly South and North American origin. These imports entail transport over long distances, conversion of natural and semi-natural habitats into arable land and raise problems of consumer acceptance.

Increasing grain legume production in Europe could be an excellent alternative to the import of soya bean meal. Thanks to their ability to symbiotically fix nitrogen from the air, grain legumes do not need any nitrogen fertilisation. Despite this feature, grain legume cultivation has been neglected within the EU for many years. This study, carried out in the concerted action GL-Pro of the EU, evaluates the impact of the introduction of grain legumes in typical crop rotations in Europe.

2. LCA CASE STUDIES

Four regions with potential for increasing their grain legume area were chosen for this study (Tab. 1): Saxony-Anhalt (Germany), Barrois (France), Canton Vaud (Switzerland) and Castilla y León (Spain). In each of these regions, two crop rotations were defined: a typical cereal-based rotation without grain legumes and an alternative rotation including grain legumes. The production data were collected by the local project partners from statistics, surveys, literature, documents from extension services and using expert knowledge [1]. The impacts of these two crop rotations were compared relative to three functional units representing different functions of agriculture: hectare per year as a measure of the land management function, € gross margin 1 for the financial function and GJ gross energy of the harvested biomass for the productive function.

The LCA calculations were carried out with the method SALCA (Swiss Agricultural Life Cycle Assessment of ART) according to [2]. The life cycle inventories were taken from the ecoinvent database version 1.2 [3].

Tab. 1: Overview of the crop rotations compared in the four study regions. GL = grain legumes, OSR = oilseed rape, W = winter wheat, wB = winter barley, sB = spring barley, P = spring peas, wP = winter peas, M = grain maize, SB = soya beans, SF = sunflowers, (cc) = catch crop (*Phacelia*). The replaced crops are shown in bold.

Region	**Crop rotation 1 (without GL)**	**Crop rotation 2 (with GL)**
Saxony-Anhalt (D)	OSR-W-**W**-W-wB	OSR-W-**P**-W-wB
Barrois (F)	OSR-W-W-wB	OSR-W-**wP**-W-wB
Canton Vaud (CH)	OSR-W-(cc)**M**-W-OSR-W-(cc)**M**-W	OSR-W-(cc)**P**-W-OSR-W-(cc)**SB**-W
Castilla y León (E)	**SF**-W-wB-sB	**P**-W-wB-sB

3. LCA RESULTS AND DISCUSSION

The full results of the study can be found in [4]. The introduction of grain legumes into intensive crop rotations with a high proportion of cereals and intensive N-fertilisation leads to a significant reduction of energy demand (Fig. 1). This reduction was achieved thanks to three factors: (i) no N fertiliser needed for the grain legume crop, (ii) less N required for the crop following the grain legume and (iii) improved conditions to reduce the soil tillage intensity. The global warming and ozone formation potentials show the same trends (Tab. 2).

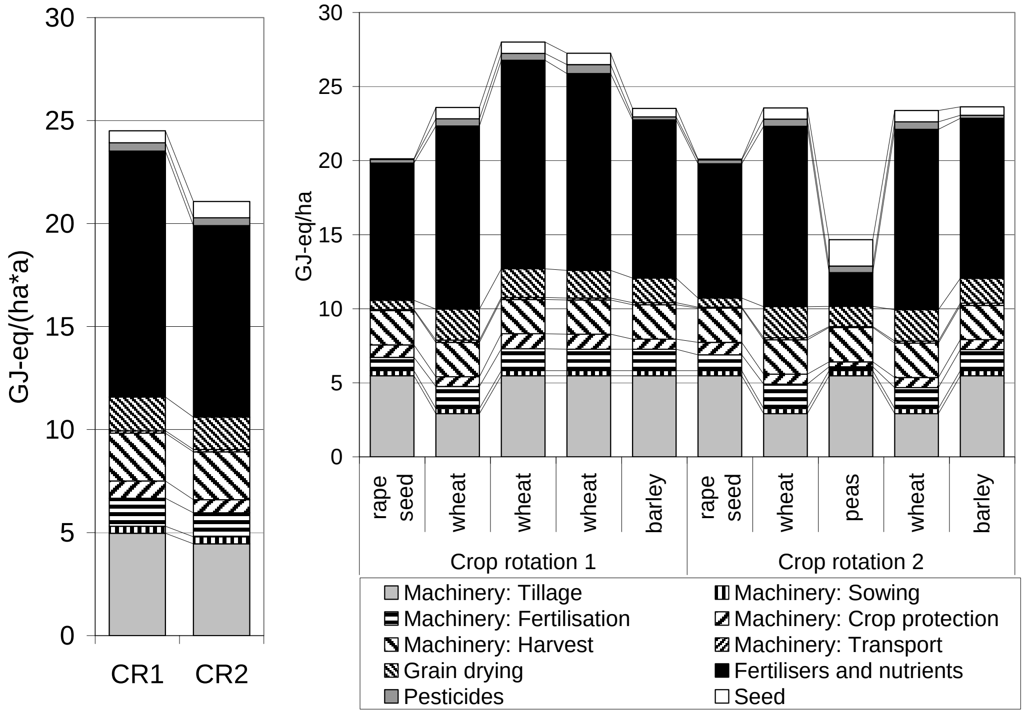


Fig. 1: Demand for non-renewable energy resources of crop rotation 1 (CR1) without grain legumes and crop rotation 2 (CR2) with grain legumes in Saxony-Anhalt (D).

The greater diversification of the crop rotation further helps to reduce problems caused by weeds and pathogens and therefore pesticide applications. Consequently, the ecotoxicity potentials are reduced in such cropping systems.

The nitrate leaching potential tends to be higher in general, but can be reduced by including catch crops or sowing winter grain legumes. No differences were found for the impacts of crop management on soil quality and biodiversity (studied in Canton Vaud only, see [4]). In the low-input crop rotation in Spain, the introduction of peas had no favourable environmental effect, mainly because little or no N-fertiliser can be saved (the intensity level is already very low). Furthermore the break-crop sunflower is replaced by another break-crop (pea). The analysis per € gross margin 1 (financial function) leads to slightly more favourable results for the grain legume crop rotations compared to the analysis per ha and year [4].

Tab. 2: Environmental impacts per hectare times year (ha*year) for crop rotations without grain legumes (CR1) and crop rotations with grain legumes (CR2). The impacts of CR2 relative to CR1 are assessed to be: ++ = very favourable, + = favourable, 0 = similar, - = unfavourable, -- = very unfavourable.

		Saxony-Anhalt (D)			Barrois (F)			Vaud (CH)			Castilla y Léon (E)		
		CR1	CR2		CR1	CR2		CR1	CR2		CR1	CR2	
Resource management	energy demand [GJ-eq]	24.5	21.1	**++**	22.5	19.9	**++**	31.5	21.9	**++**	10.3	10.7	**0**
	global warming potential [t CO2-eq]	3.76	3.33	**++**	3.97	3.67	**+**	4.00	3.65	**+**	1.95	2.17	**-**
	ozone formation [g C2H4-eq]	790	709	**+**	669	629	**+**	854	728	**++**	335	354	**-**
Nutrient manage-ment	eutrophication [kg N-eq]	48.2	47.4	**0**	100.9	94.7	**0**	58.8	64.4	**-**	63.4	72.8	**-**
	acidification [kg SO2-eq]	21.4	17.7	**+**	44.4	36.3	**+**	20.4	17.5	**+**	9.4	9.8	**0**
Pollutant management	terrestrial ecotoxicity EDIP [points]	50929	32293	**++**	11413	10603	**0**	731	862	**-**	387	401	**0**
	aquatic ecotoxicity EDIP [points]	3846	3904	**0**	4701	4088	**+**	2708	2611	**0**	3335	2471	**+**
	terrestrial ecotoxicity CML [points]	194	193	**0**	1095	878	**+**	689	691	**0**	8	11	**-**
	aquatic ecotoxicity CML [points]	595	571	**0**	2742	2219	**+**	2097	2375	**-**	104	93	**0**
	human toxicity CML [points]	747	636	**+**	990	856	**+**	1334	1261	**0**	328	342	**0**

Due to the lower yields of grain legumes compared with cereals, the advantages of grain legumes are smaller when considered per GJ gross energy of the harvested products (productive function, Tab. 3). However, the energy efficiency is higher in crop rotations with grain legumes.

4. CONCLUSIONS

Based on these four case studies we conclude that the introduction of grain legumes into intensive crop rotations with a high proportion of cereals and intensive N-fertilisation is likely to reduce energy use, global warming potential, ozone formation and acidification as well as eco- and human toxicity per unit of cultivated area. The nitrate leaching potential tends to be higher with grain legumes in general, but can be reduced by including catch crops or sowing winter grain legumes. In low-input crop rotations like the one in Spain, no significant changes in environmental impacts are to be expected, especially when a break-crop is replaced by a grain legume. From an environmental point of view, the inclusion of grain legumes in European crop rotations offers the potential to reduce environmental burdens.

5. ACKNOWLEDGEMENTS

This research was supported by the European Commission (grant no. QLK5-CT-2002-02418) and by the Swiss State Secretariat for Education and Research.

Tab. 3: Environmental impacts per GJ gross energy of the harvested products for crop rotations without grain legumes (CR1) and with grain legumes (CR2). The impacts of CR2 relative to CR1 are judged to be: ++ = very favourable, + = favourable, 0 = similar, - = unfavourable, -- = very unfavourable.

		Saxony-Anhalt (D)			Barrois (F)			Vaud (CH)			Castilla y Léon (E)		
		CR1	CR2		CR1	CR2		CR1	CR2		CR1	CR2	
Resource management	energy demand [kJ-eq]	227	210	+	233	217	+	294	251	++	256	268	-
	global warming potential [g CO2-eq]	35	33	+	41	40	0	37	42	--	48	54	-
	ozone formation [g C2H4-eq]	7.3	7.1	0	6.9	6.8	0	8.0	8.4	-	8.3	8.8	-
Nutrient manage-ment	eutrophication [g N-eq]	446	471	0	1046	1030	0	547	740	--	1569	1817	-
	acidification [g SO2-eq]	199	176	+	460	395	+	190	201	0	232	244	0
Pollutant management	terrestrial ecotoxicity EDIP [points]	472	321	++	118	115	0	7	10	--	10	10	0
	aquatic ecotoxicity EDIP [points]	36	39	0	49	44	0	25	30	-	82	62	+
	terrestrial ecotoxicity CML [points]	1.8	1.9	0	11.3	9.5	+	6.4	7.9	-	0.2	0.3	-
	aquatic ecotoxicity CML [points]	5.5	5.7	0	28.4	24.1	+	20	27	--	2.6	2.3	0
	human toxicity CML [points]	6.9	6.3	0	10.3	9.3	0	12.4	14.5	-	8.1	8.5	0
	Gross energy production GJ/(ha*year)	108.0	100.5		96.5	91.9		107.4	87.0		40.4	40.1	

6. REFERENCES

1. von Richthofen, J.S., Pahl, H., Nemecek, T., Odermatt, O., Charles, R., Casta, P., Sombrero, A., Lafarga, A., Dubois, G., 2006. Economic interest of grain legumes in European crop rotations. GL-Pro report, WP3.
2. Nemecek, T., Huguenin-Elie, O., Dubois, D., Gaillard, G., 2005. Ökobilanzierung von Anbausystemen im schweizerischen Acker- und Futterbau. Agroscope FAL Reckenholz, Zürich; Schriftenreihe der FAL 58, 155 p.
3. Frischknecht, R., Jungbluth, N., Althaus, H.-J., Doka, G., Hellweg, S., Hischier, R., Nemecek, T., Rebitzer, G., Spielmann, M., 2004a. Overview and Methodology - ecoinvent data v1.1. Swiss Centre for Life Cycle Inventories (ecoinvent), Dübendorf; ecoinvent report 1, 75 p.
4. Nemecek T. & Baumgartner D., 2006. Environmental Impacts of Introducing Grain Legumes into European Crop Rotations and Pig Feed Formulas. Concerted Action GL-Pro, Final report WP4. ART, 63 S. http://www.art.admin.ch/themen/00617/00622/index.html?lang=de.

Improving the ecoefficiency of *orxata* production. Preliminary LCA of *orxata* manufacturing process

N. Sanjuán[a], J. Ribal[b], C. Enguix[c], L. Fenollosa[b], G. Clemente[a]

[a]ASPA Group. Department of Food Technology. Universidad Politécnica de Valencia, 46021 Valencia, Spain.
[b]Departamento of Economy and Social Sciencies. Universidad Politécnica de Valencia, 46021 Valencia, Spain.
[c]AINIA, Agrofood Technological Institute, Valencia Parc Tecnologic, 46890 Paterna, Spain

Abstract
Orxata is a traditional soft drink of Valencia made from Tigernuts *xufa* (*Cyperus sculentum*) tubers. The results of a streamlined LCA on the present production practices are shown in this study. Two systems representative of the manufacturing process have been taken into account. From the results it can be highlighted that the stages with a higher impact are the refrigeration of the orxata and the cleaning of the processing plant after the manufacturing process. Other production scenarios that will be evaluated in the future from the economic and environmental point of view have been proposed.

1. INTRODUCTION

Xufa (*Cyperus sculentum* L.) is a wild herbaceous plant, which is also cultivated to industrialize its tubers. In Spain it is mainly grown in the Comunitat Valenciana, specifically in the Horta Nord an area at the north of Valencia city.

The tubers (tiger nuts, chufas or *xufas*) have a high fat content and are used in some countries to extract oil. In Spain tiger nut production is mainly processed to obtain *orxata* (horchata in Spanish), a traditional soft drink.

Orxata is obtained after soaking and washing the tubers in water, afterwards they are milled to obtain a paste, to which water is added, then filtered and, finally, the liquid is sweetened with sugar. Although nowadays it is subsequently processed in different ways (concentrated, sterilised, frozen…) in order to increase the self life of the product, traditionally *orxata* is refrigerated after processing, obtaining the so called *orxata natural* that usually is directly sold to the public in the same establishment.

For this reason, cleansing of both the tubers and the equipment are crucial steps of the process, which demand high water consumption and the use of germicides. Nevertheless, when improving the processing system, environmental aspects, such as the consumption of water and electricity, should also be taken into account, without forgetting the production costs (Stefanis *et al*., 1997; Azapagic and Clift, 1999).

This study is in keeping with a project that aims to develop a methodology that helps us to incorporate the best available techniques in food processes incorporating technical, economic and environmental aspects.

The goal of this study is to detect the most impacting practices of *natural orxata* production process by using a streamlined LCA and to propose alternative production scenarios, which will be further evaluated.

2. METHODOLOGY

Two basic semicontinuos existing processes (A and B) for obtaining *orxata* were assessed (figure 1). Main differences between the two processes lie in:

- washing and soaking of tigernuts, that takes 5-8 h in process B and 50 min in process A.
- *orxata* extraction equipment, which consist of a blade mill (15 CV) and mixer-filter in A and a hammer mill (4 CV), a press and a rotary filter in B.
- a second extraction of liquid is made from the solid waste in process B.

Taking into account that the shelf life of this kind of *orxata* is 3 days at 2ºC, a refrigeration period of two days was supposed.

Four modules were considered in the LCA: tigernuts cleansing, tigernuts processing, refrigeration of the final product and cleaning of the facilities after processing.The functional unit considered was 300 L of refrigerated *orxata*. Data have been provided by two companies.

A streamlined LCA was performed (Todd and Curran, 1999), for this reason upstream and downstream processes were omitted and the impact categories considered were: water (L) and electricity (MJ) directly consumed in the process, eutrophication (g eq. PO_4^{3-}) and global warming (g eq. CO_2). Due to lack of data neither the manufacturing of sanitizers and cleaning agents neither the inherent toxicity impacts caused by these products have not been considered in the study.

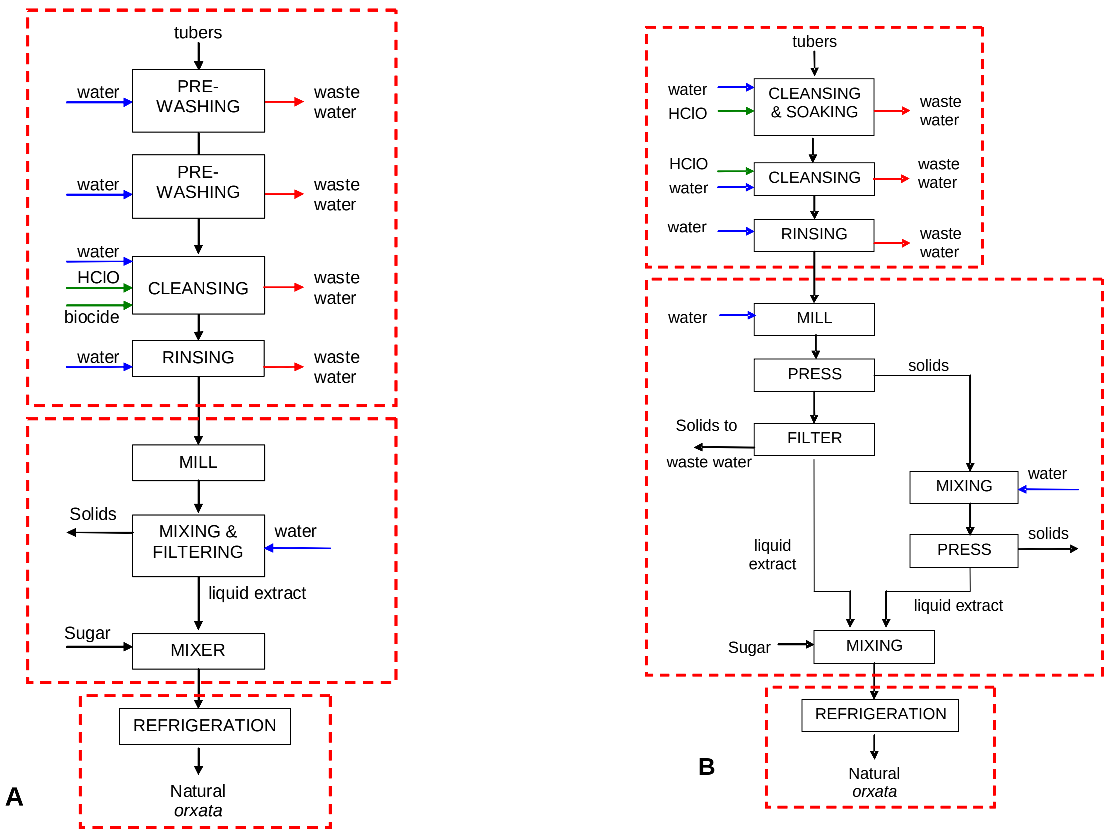


Figure 1. Flow chart of the *orxata* production process.

3. RESULTS

Figures 2, 3, 4 and 5 show the results obtained for process A and Bb in the impact categories studied. With respect to water, refrigeration is the stage demanding the highest consumption (Figure 2). Process A requires 31% more water than B, mainly due to the cleansing of the tubers. The amount of water added due the processing stage is higher in A (270 l) than in B (200 l) since tigernuts are not rehydrated in process A.

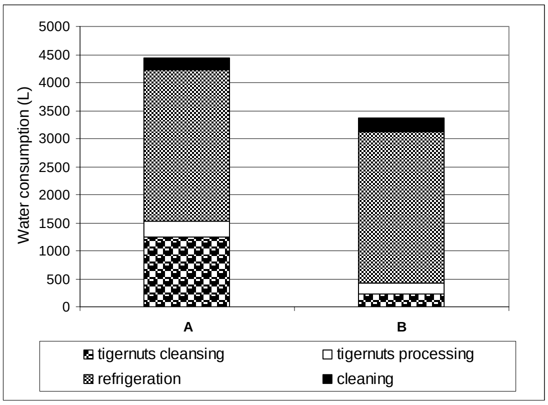


Figure 2. Water consumption (l).

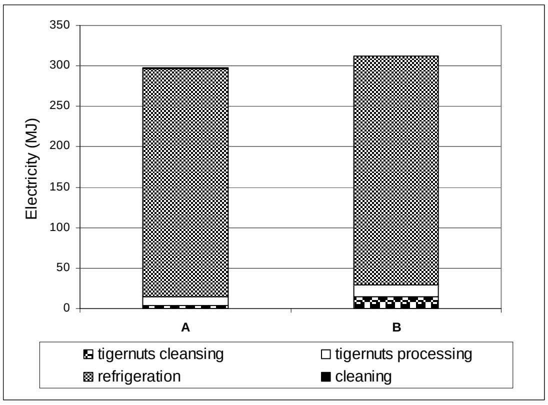


Figure 3. Electricity consumption (MJ).

With regard to the electricity consumption, it is a 4.8% higher in process B. Differences between the two processes are due to the cleansing and processing of the tubers. As in the water consumption, the refrigeration is the stage with the highest electricity consumption.

Since the only source of greenhouse emissions is the electricity consumption, the results obtained for the greenhouse effect (figure 4) follow the same pattern.
With respect to the eutrophication potential (figure 5), in both processes the main contributor to this impact is the cleaning of the equipment after *orxata* processing. Another critical stage is the cleansing of the tubers.

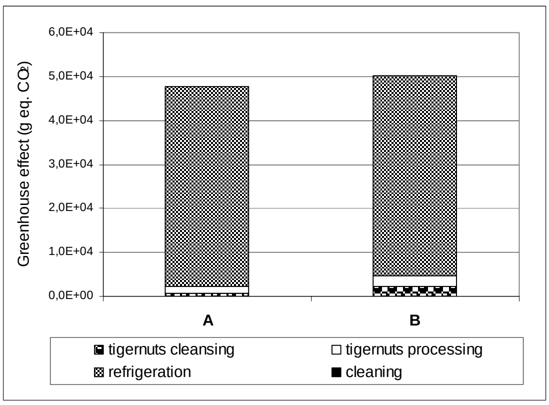


Figure 4. Greenhouse effect (100 years).

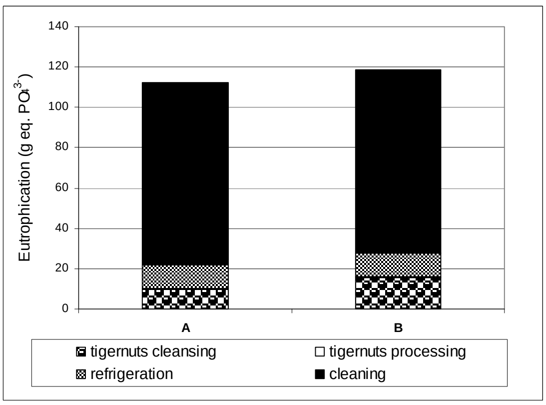


Figure 5. Eutrophication.

From the results it can be highlighted the refrigeration of the *orxata* drink as a crucial step in the studied life cycle due to is high water and electricity demand. Another stage that must be taken into account is the cleaning of the processing plant.
Thus, the aspects to be taken into account in the scenarios to be evaluated are:

a) tigernuts cleansing:

- cleansing of the tigernuts by using ozone + recycling of rinsing water
- minimization of the amount of soaking water

b) refrigeration system:

- recycling of refrigeration water
- change of the refrigeration tanks to tanks without water circuit

c) cleaning of the facilities:

- dry cleaning at the beginning of the cleaning process
- reuse of tigernuts cleansing water as pre-cleaning water
- reuse of water of the refrigeration tank as pre-cleaning water
- enzymatic cleaning of the facilities

To evaluate the change to other kind of cleaning products will imply to include in the assessment the toxicity impacts.

All scenarios will be environmentally and economically assessed, taking into account technical and microbiological quality aspects. In order to find the best scenario, Data Envelopment Assessment (DEA) techniques will be used. DEA will allow to estimate a unique measurement of ecoefficiency for every scenario. One of the main advantages of DEA techniques is that allocates objective weights to the environmental impacts.

4. CONCLUSIONS

From the streamlined LCA it can be concluded that both the refrigeration of the *orxata* and the cleansing of the facilities after processing are critical stages of the process. Regarding the tubers processing, the two studied systems (A and B) have a similar impact. Alternative scenarios to decrease the environmental impact of the process have been proposed which will be evaluated from the environmental and economic point of view.

5. ACKNOWLEDGEMENTS

The authors acknowledge the support from IMPIVA ref. IMCOCA/2006/57

6. REFERENCES

Azapagic, A. and Clift R. Life cycle assessment and multiobjective optimisation. 1999. *Journal of Cleaner Production* 7, 135-143.

Stefanis, S.K., Livingston, A.G. and Pistikopoulos, E.N. 1997. Environmental impact consideration in the optimal design and scheduling of batch processes. C*omputers and Chemical Engineering* 21(19), 1073-1094.

Todd, J.A. and Curran, M.A. (Ed.). 1999. *Streamlined Life Cycle Assessment: a final report from the SETAC North America Streamlined LCA workgroup*. Society of Environmental Toxicity and Chemistry (SETAC), Pensacola, FL, USA. Available at: www.**setac**.org/files/**lca**.pdf.

Food Product LCAs as Evidence to Support Policy-making

Chris Foster, Ken Green, Paul Dewick.
Manchester Business School, UK

Abstract
A review project found that LCA has been applied to the primary production end of product life cycles for many food commodities produced in Europe, with considerable agreement between studies. These studies are supported by research into the processes that link emissions from agricultural processes to changes in inputs. Some food LCAs also show that other stages of food-product life cycles, for example domestic cooking and wastage, should not be neglected in policy programmes aimed at reducing the overall environmental impact of food production and consumption. Rather few studies highlight the increases in burdens that arise as more processing is introduced into food product systems, although this is a continuing trend linked to the drive to "add value". We conclude that, while further development of the evidence base about agricultural production is worthwhile, if food product LCAs are to inform food policy, future research must encompass the post-farm stages of food product life-cycles.

INTRODUCTION

The environmental significance of food production has been recognised for a long time, and should come as no surprise: agriculture is after all a direct, planned intervention by man in the "natural environment" that takes place on a significant fraction of the Earth's surface. The EIPRO study (Tukker *et al* 2005), undertaken to inform the EU's Integrated Product Policy initiative, stressed the relative importance of the wider food production and consumption system as a contributor to the overall environmental burdens of European life. This and related evidence leads policymakers to ask 3 questions:

- What gives rise to these impacts?
- How do current trends in food production and consumption affect the pattern of impacts?
- Where can actions reasonably be taken to reduce the environmental impact of food production and consumption?

FOOD PRODUCT LCAs

In an attempt to address these questions, a review of food product LCAs was carried out in 2005-6 by a team from Manchester Business School (Foster, Green *et al* 2006) to ascertain to what extent this body of work answered these questions. We found that:

There are many methodological issues relating to the application of LCA to foods; ways of tackling a number of these have been developed. This development has focussed on the primary production end of the food chain - the modelling of agricultural production systems and the generation of good-quality LCAs for basic food commodities (see, for example Williams et al 2006, Danish LCA Food Database). Good agreement is beginning to emerge from this work, for example results for the impact of milk production in the UK and Sweden are similar, with differences being at an understandable and explicable level. So far, this body of work has mostly considered European production but not production in locations further afield commonly used to source food for European processing and consumption.

The application of LCA to food products highlights limitations of common practice in terms of Life Cycle Impact Assessment. Land use, the impact of water use on local resource bases and the ecotoxicological effects of pesticides are impacts of food production that are of concern to many but for which LCIA methods are still under development (e.g. within the UNEP-SETAC Life Cycle Initiative) and/or little used.

Reflecting the focus of methodological development, the coverage of food products by published LCA studies is weighted heavily towards food commodities, fruit, vegetables and simple finished products derived from few ingredients (for example bread and cheese). The results of LCAs of these types of finished products show that agricultural production and production of agricultural inputs are the dominant source of environmental impacts. (There is a tendency to bring these two parts of food product systems together when reporting LCA results, which rather obscures the sources of impacts). The few studies that have examined more complex products show that processing can add significantly to the overall burdens in the system. For example Andersson *et al.* (1998) found processing to account for over 30% of the total global warming potential associated with tomato ketchup, Feitz *et al. (2005)* have drawn attention to the greatly increased energy demand arising from production of added value dairy products and Ligthart *et al.* (2005) have demonstrated the additional burdens introduced by freezing vegetables.

The system boundaries in few food LCAs extend as far as the final consumer, however. Studies of commodities or simple products stop, understandably, at the farm gate or the factory's store. Only the few studies investigating more complex products take into account the actions of the consumer. But it is clear from the work of Carlsson-Kanyama (e.g. in Carlsson-Kanyama and Boström-Carlsson (2001)), from Sonesson et al (2005a & b), from Pretty *et al* (2005), from our own calculations and the work of WRAP (2007) that what consumers do with food, in transporting it, cooking it or indeed failing to utilise it at all, critically influences the overall impacts of food product systems.

DISCUSSION

The food product LCAs undertaken to date partially answer the first question posed in the introduction, highlighting and explaining the environmental impact of primary food production and the production of agricultural inputs. Published LCA studies of more complex, or more highly processed, food products are too few in number to allow the environmental significance of food processing to be properly evaluated.

As for the second question, food LCAs suggest that while converting food production to organic methods may bring some environmental benefits, it won't have a very large effect for some key environmental themes (notably greenhouse gas emissions and eutrophication) and might even bring increases in some impact areas. The very few studies that have used LCA to investigate the relative environmental impacts of local food systems compared to more global ones provide no substantial evidence for clear environmental benefits from the former, while studies of single-product systems suggest that bulk transport of foodstuffs is seldom one of the more environmentally- significant stages of the life cycle (with the possible exception of air-freighted produce).

There are too few published studies to allow us to understand environmental implications of the shift of consumption towards food products in more convenient formats. In the light of work on more complex products and extending as far as the final consumer, it appears that wastage rates at different points in the processing chain (including the household) and the performance of domestic cooking equipment are likely to be critical parameters in any such comparison. Given the energy-intensity of the drying process (see for example Feitz *et al* 2005), the extent to which convenience foods incorporate dried components may also be an important factor. Although it was not within our remit, we encountered no LCA work directed at food products delivered via foodservice, although this too is an important growth area for the industry.

With respect to the third question in the introduction, food LCAs don't offer many new additions to the existing box of tools available for the "greening" of agricultural production, although they highlight some of its limitations. They also remind policymakers that larger-scale operations are almost always more resource-efficient than smaller-scale ones, suggesting that any economic and social benefits associated with "local" food systems may well be gained at some environmental cost. Taken overall, food LCAs also lead to the view that reducing the environmental impacts of food production and consumption in absolute terms is likely to require policy measures that eventually affect consumers' behaviour, even if the available evidence about consumers' response to environmental information suggests that the best way to do that will probably not involve engaging them as the primary agents of change.

CONCLUSIONS

The food LCAs undertaken to date provide some help to policymakers, but the existing body of work (which we recognise as being relatively recent, at least partly explaining its limited scope) opens up as many questions as it closes. On the back of our research and other recent work, we suggest that:

More LCAs of processed food items are needed. These should extend as far as the final consumer: it would be helpful if some consensus emerged about handling the methodological challenges that will emerge in the process. LCAs of food commodities produced in non-European countries are needed to allow further investigation of the environmental balance between distant and local sourcing. LCAs need to take account of different utilisation of different food products.

Studies of the environmental impacts of food need to consider environmental themes not easily covered by LCA, whether they do this by incorporating new development in Impact Assessment or by mixing LCA and non-LCA methods while retaining a single frame of reference (i.e. the product system).

Methodological issues will arise if we try to use LCA to understand whether the undoubted additional burdens of some food processing (freezing and drying, for example) bring a net benefit to society. Challenges will revolve around evaluating and defining the functionality of food products (frozen carrots clearly have a different functionality from fresh carrots which derives from their availability to the user over an extended time period), designing studies to recognise that many food product systems work with a fixed and perishable input, obtaining and incorporating good data about the consumer end of the system, as well as the more familiar problem posed by the existence of multi-product processes throughout the food system.

REFERENCES

K. Andersson, T. Ohlsson & P. Ohlsson, (1998).
Screening LCA of tomato ketchup. Journal of Cleaner Production **6** pp 277 – 288.
A. Carlsson-Kanyama and K. Boström-Carlsson (2001)).
Energy Use for Cooking and Other Stages of the Life Cycle of Food. FMS No. 160 report. Stockholm University.
Danish LCA Food Database (2003 or later) www.lcafood.dk
A. Feitz, S. Lundie, G. Dennien, M. Morain, and M. Jones, (2005).

Comparing agricultural crops for bio-product applications

M.A. Renouf

School of Geography, Planning and Architecture, The University of Queensland, QLD 4072, Australia

Abstract

An LCA case study is presented that compares sugarcane, corn and sugar beet. The comparison was made on the basis that each crop produces a sugar solution that can be used in the production of fermentation bio-products. The results show that sugarcane provides advantages in relation to some impacts, but that sugar beet provides advantages in relation to others. The factors found to strongly influence the relative performance of the crops were yield, the nature and quantities of commodities displaced by co-products, and field emissions (particularly nitrogen).

The paper discusses the difficulties associated with comparing crops, including the representation of variability and assumptions made about product displacement. It also questions whether the direct comparison of agricultural crops is valid, given their regional specificity.

1. INTRODUCTION

The environmental evaluation of agricultural crops as sources of bio-products is an area of growing interest. Past assessments have commonly applied life cycle assessment, but with a focus on energy input and greenhouse. However recent work has included other impacts such as eutrophication and acidification potential. These advances, along with the concerted efforts by LCA practitioners to refine methodologies for agriculture generally, have laid a solid base for the environmental characterization of agricultural crops as sources of bio-products. In this paper, the use of LCA for comparing crops as substrates for bio-products is discussed.

2. CASE STUDY – COMPARISON OF THREE SUGAR-PRODUCING CROPS

LCA was used to compare the environmental profiles of three sugar-producing crops – sugarcane, corn and sugar beet. The sugarcane system was based on practices in the state of Queensland, Australia. The corn analysis was based on data from the United States [1, 2], and the sugar beet analysis was based on data from the United Kingdom [3, 4].

The comparison was made on the basis that each crop produces a functionally equivalent product – a sugar solution containing mono-saccharide of similar sugar purity – and focused on the agronomic and processing characteristics of the crops and not on factors related to where they are grown. The processes for extracting a sugar solution from each crop are different, but the output common to all is a clarified sugar solution, which was taken as the reference product (Figure 1). The functional unit is sugar solution containing a tonne of mono-saccharide equivalent. Each crop produces different co-products, which were accounted for using system expansion.

The results show that sugarcane has a discernable advantage in relation to energy input (Figure 2) and greenhouse gas emissions (Figure 3). This is due to the availability of surplus bagasse fibre which can be used as a renewable fuel, generating 'credits' from the displacement of fossil fuels.

Sugar beet appears to have an advantage in relation to eutrophication (Figure 4) and a shared advantage with sugarcane in relation to acidification potential (Figure 5). Sugar beet has a slightly lower N input than the other crops and correspondingly lower potential for nitrogen loss. The eutrophication and acidification impacts of its production are also offset by the avoided production of other agricultural crops displaced when sugar beet pulp is used as an animal feed.

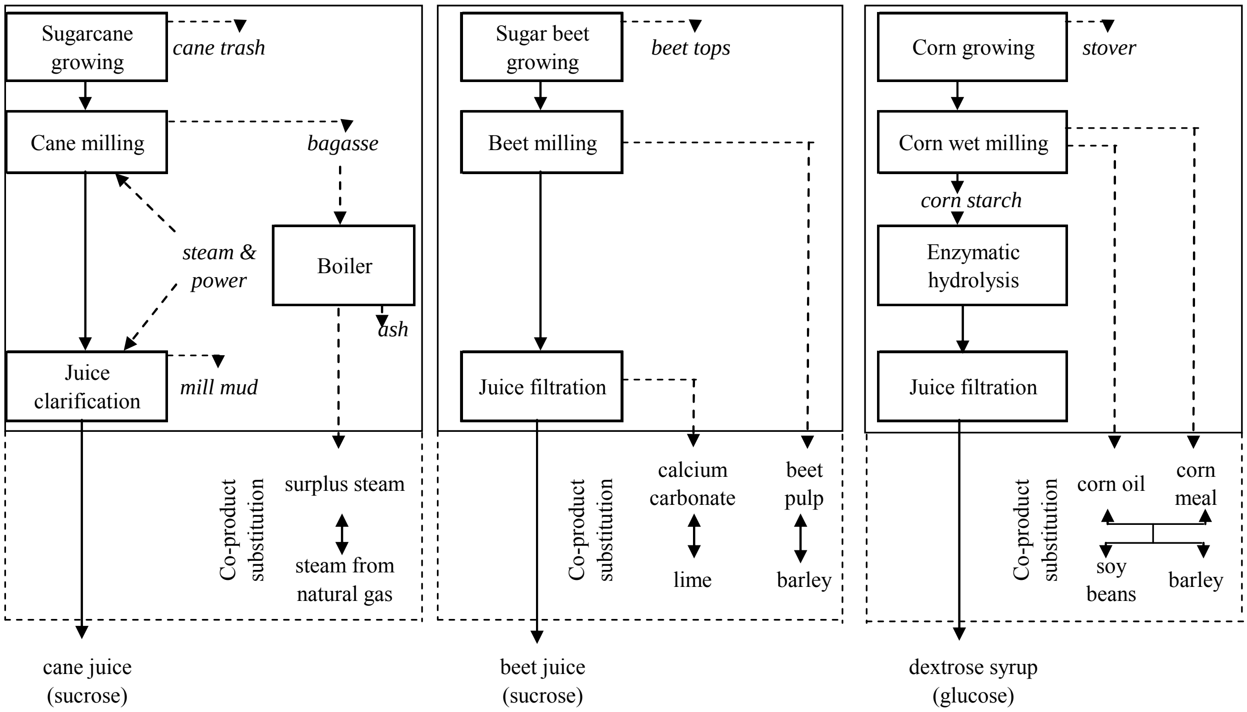


Figure 1.—Production of saccharide from sugarcane, corn, and sugar beet.

The corn system can also claim credits for the displacement of other crop production with it co-products (corn oil and meals). However in this analysis the credits were not sufficient to offset the higher impacts or corn production due to its low saccharide yield relative to the other crops.

A disadvantage of sugarcane is its potentially high water use (400kL/tonne saccharide on average), compared with the other crops, which require very little. High water use is often required to achieve the high sugar yields required for economically viable production.

A factor contributing to sugarcane's potential advantages is its high yield. It is a very efficient crop at capturing solar energy and converting it into carbohydrate. This also means that sugarcane demands less agricultural land than the other two crops to produce a tonne of saccharide (0.08ha for sugarcane, compared with 0.18ha for corn and 0.13ha for sugar beet).

3. INFLUENTIAL FACTORS

Three factors strongly influenced the results; agricultural yield, the nature and quantities of commodities displaced by co-products and field emissions. The importance of agricultural yield is demonstrated most strikingly for corn. While all crops have similar inputs and outputs per hectare, corn is disadvantaged by its low yield of saccharide (5.5 t/ha), compared with sugarcane (11.7 t/ha) and sugar beet (7.6 t/ha).

Sugarcane provides energy and greenhouse benefits because it displaces fossil-fuels, whereas sugar beet and corn can reduce eutrophication and acidification effects because they displace other agricultural production. The nature and quantity of the commodities displaced by the co-products determine the environmental credits that can be assigned.

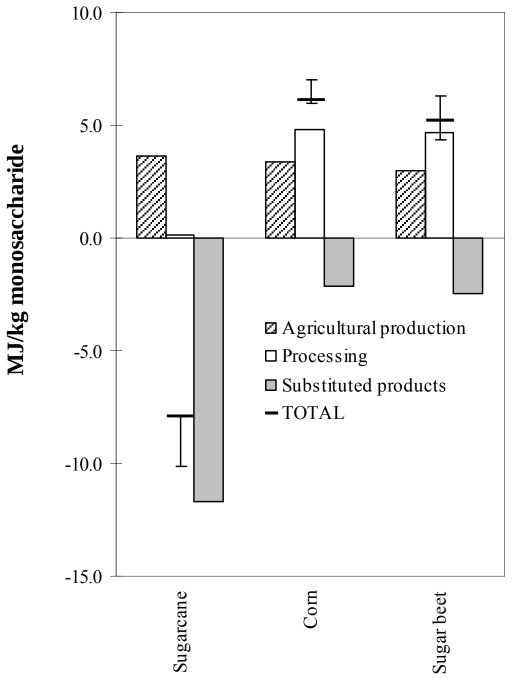


Figure 2. Energy input results.

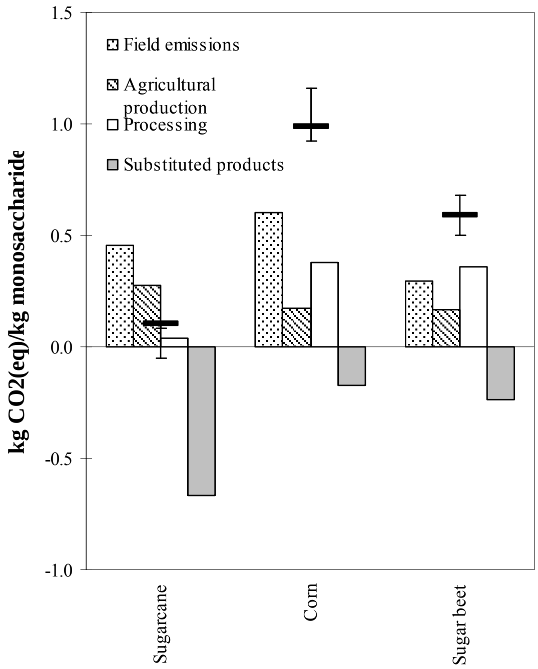


Figure 3. Greenhouse gas results.

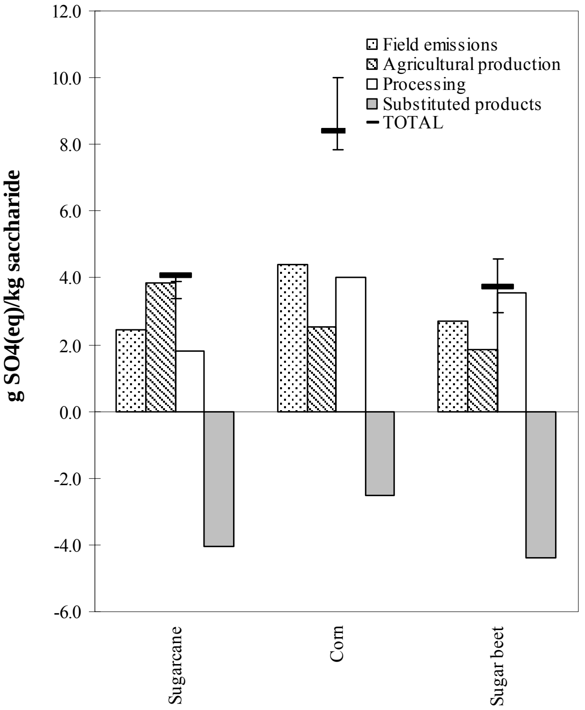


Figure 4. Eutrophication potential results.

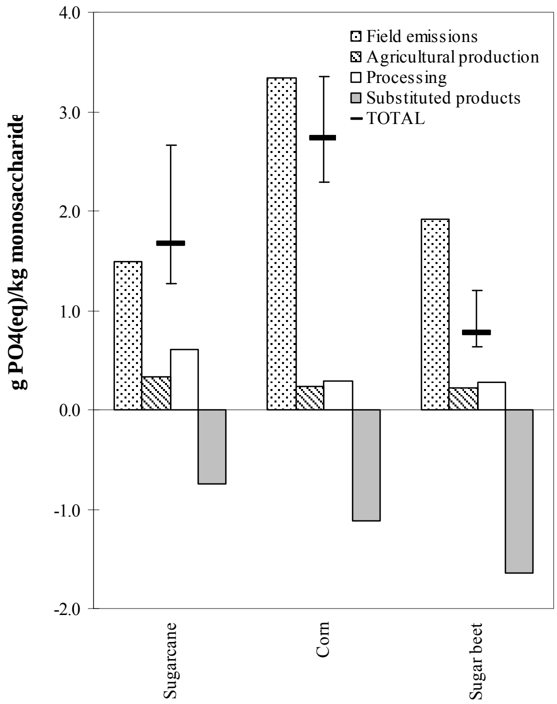


Figure 5. Acidification potential results.

Field emissions, particularly of nitrogen species, have an important influence on the results. Field emissions are strongly influenced by site-specific factors and their management at the local level will greatly influence the results. Field emissions are difficult to account for accurately, due to the complex interactions between the soil, the crop, and the surrounding environment. So as well as being an important aspect, it is also the aspect of greatest uncertainty.

4. DIFFICULTIES COMPARING CROPS

4.1 Representing variability

Understanding and representing variability is important for agricultural systems [5, 6]. At the outset, agricultural inputs per tonne of product were assumed to be a good predictive indicator of environmental impacts, and were used to select regions that could be expected to represent high- and low-impact scenarios. However this approach did not lead to an accurate representation of variability (as seen in the error bars for sugarcane in Figure 2 to 5). In the end, environmental conditions (climate, soil type, etc.) and agronomic practices were found to be more influential.

Variability in environmental conditions may be the most important factor, as it has a strong influence on field emissions as well as agricultural inputs. Agronomic practices, dictated by grower preference, also appear to have an important influence, but this aspect was not fully explored in the study. Geographic location relative to supporting infrastructure was not found to be highly influential, since the environmental aspects it influences (transportation and origin of agro-chemical inputs) were not found to be significant.

Careful consideration, including pre-screening, needs to be given to the representation of variability. Environmental conditions (climate, soil type, topography, etc.) may need to be the most important factor to consider when representing ranges.

4.2 Assumption about product displacement

The analysis made assumptions about the product displacement effects of co-products in the Australian context. Others have assigned larger co-product credits to corn than in this work by undertaking a more expansive assessment of the displacement impacts of corn co-products in the US context [7]. The influence of product displacement assumptions is thought to be great when comparing different agro-industrial systems and needs to be further explored.

4.3 Regional-specificity of crops

While the case study addressed an interesting question, “how does sugarcane compare with other potentially competing sugar-bearing crops?” it begs the question, “is it an appropriate comparison”? Sugarcane, corn and sugar beet are each suited in different climatic conditions. Can we directly compare crops considering that in most cases “you grow what you can where you can”? Most crops are suited to specific climates and conditions and are not directly substitutable.

The case study identified that field emissions were a dominant contributor to many of the environmental impacts assessed. Since field emissions are strongly influenced by site-specific conditions, it may not be valid to compare crops without considering the location in which it is grown. Therefore it may be more appropriate to compare agro-industrial systems in the regional context in which they occur, data permitting.

5. ACKNOWLEDGEMENTS

This work was part of a PhD at The University of Queensland (UQ), Australia. Funding was provided by an Australian Postgraduate Award and the Cooperative Research Centre for Sugar Industry Innovation through Biotechnology. The author would like to acknowledge the contribution of Dr. Malcolm Wegener and Assoc. Prof. Lars Nielsen, also from UQ.

6. REFERENCES

[1] Kim S, Dale BE. Cumulative energy and global warming impact from the production of biomass for biobased products. Journal of Industrial Ecology 2004; 7: 1147-162.

[2] Kim S, Dale BE. Life cycle assessment of various cropping systems utilized for producing biofuels: bioethanol and biodiesel. Biomass and Bioenergy 2005; 29: 426-39.

[3] Tzilivakis J, Warner DJ, May M, Lewis KA, Jaggard K. An assessment of the energy inputs and greenhouse gas emission in sugar beet (Beta vulgaris) production in the UK. Agricultural Systems 2004; 85: 101-19.
[4] Mortimer ND, Elsayed MA, Horne RE. Energy and greenhouse gas emissions for bioethanol production from wheat grain and sugar beet. Final Report for British Sugar plc.: School of Environmental and Development, Sheffield Hallam University; 2004.
[5] Ferret R, Mendoza G, Castilla M. The Influence of Agricultural Data Uncertainty in the Life Cycle Assessment of Biodegradable Hydraulic Lubricants. In: Pahl-Wostl C, Schidt S, Rizzoli AE, Jakeman AJ, editors. Complexity and Integrated Resource Management. Transactions of the 2nd Biennial Meeting of the International Conference of the International Modelling and Software Society, University of Osnabruck, Germany; 2004.
[6] Halberg N, Kristensen IS, Dalgaard T. Linking data sources and models at the levels of processes, farm types and regions. In: Weidema BP, Meeusen MJG, editors. Agricultural data for life cycle assessment. The Hague: Agricultural Economics Research Institute; 2000, p. 16-30.
[7] Kim S, Dale BE. Allocation procedure in ethanol production system from corn grain. International Journal of Life Cycle Assessment 2002; 7: 237-43.

Using European grain legumes in meat production: How can environmental impacts be moderated

Daniel Baumgartner[a], Thomas Nemecek[a], Julia-Sophie von Richthofen[b], Katell Crépon[c]

[a]Agroscope Reckenholz-Tänikon Research Station ART, CH-8046 Zurich, Switzerland, daniel.baumgartner@art.admin.ch
[b]proPlant GmbH, Münster, Germany
[c]UNIP, Paris, France

Abstract
Animal production is an important contributor to environmental impacts by agriculture. Three case studies have been performed to assess the environmental benefits from substituting South American soya beans by European grain legumes at two analysis levels: feedstuff and meat production. The effects are exemplified by results on energy demand for pig feeds, on eutrophication for chicken feeds and on global warming potential for pig production. The substitution of soya beans from overseas by European grain legumes in feedstuffs decreases the environmental impacts, mainly for the resource management. The environmental impacts of meat production can be moderated by reducing transport of feed ingredients and optimising cultivation practices (enhancing eco-efficiency) as well as improving agricultural and nutritional characteristics (e.g. stable and higher yields, higher protein contents in grain legumes) of the feed ingredients.

1. INTRODUCTION

Today Europe is highly dependent on imports of plant proteins for animal feeding, which mainly consists of soya beans from North and South America. This involves long transport distances as well as conversion of natural and semi-natural habitats into arable land. In combination with cropping GM-varieties, this faces problems of consumers' acceptance [1].

Cultivation of more grain legumes in Europe could be an interesting alternative, considering additionally the fact that grain legumes - being able of symbiotic nitrogen fixation - do not need any nitrogen fertilisation. In the present study, which has been performed in the frame of the integrated project GLIP of the EU, we evaluate the environmental impacts of replacing South American soya bean meal by European grain legumes on the level of feedstuffs and of animal products by means of case studies.

2. LCA CASE STUDIES

The case study region is North-Rhine Westphalia (NRW) in Germany for pig production and Brittany (BRI) in France for chicken production. These regions have been selected because of their national importance in producing the corresponding animal products [2]. The production data for the feed ingredients are taken from [3] and from the SALCA (Swiss Agricultural Life Cycle Assessment) database by ART [4]. The life cycle inventories are taken from the ecoinvent database version 1.2 [5]. The LCA calculations were performed as described in [6].

2.1 LCA of feed formulas

For pigs a three-phase feeding system has been chosen with two alternatives: feed formulas based on soya bean meal and cereals (SOY) and feed formulas based on peas, rape seed meal, soya bean meal and cereals (GLEU = grain legumes of Europe). The soya beans are from Brazil, the other feed ingredients are of German origin.

Three types of feed have been assessed for chicken production with three alternatives: SOY, GLEU and SAA (synthetic amino acids). The first two are similar to the formulas described above for pigs

with exception that more ingredients (e.g. field beans, maize, sunflower meal, palm oil) are included in the feeds and - in addition to soya beans - palm oil and a part of the sunflower meal used are from overseas production. In the SAA alternatives the required digestibility of amino acids of the formulas is increased and consequently, the feed formulas contain higher quantities of synthetic amino acids.
The system boundary is at the feed mill gate and the functional unit is 1kg of feed.

2.2 LCA of animal products

Compared to the LCA of feed formulas the system is enlarged by the animal production facilities and the resource use and emissions involved. This analysis level is illustrated for pig production in North-Rhine Westphalia. The feeds employed are the same as described under 2.1. An additional scenario named FARM is assessed using the same feed formulas as in GLEU, but without transports for feedstuffs, as the feed ingredients are produced on-farm. The functional unit is 1kg pig (live weight) and the system boundary is set at the farm gate.

3. LCA RESULTS AND DISCUSSION

3.1 Energy demand for pig feed

The production of the raw materials is the main contributor to the impact category energy demand. The total impact for the production of these materials is similar for SOY and GLEU for all three-phase feeds (Fig. 1). The energy demand of the GLEU alternatives is reduced compared to the SOY formulas. The main difference between the GLEU and SOY alternatives results from transport. Soya is increasingly produced in the vast interior parts of Brazil, which implies long transport distances by lorry to the river ports followed by a long water transport by barge and transoceanic vessel.

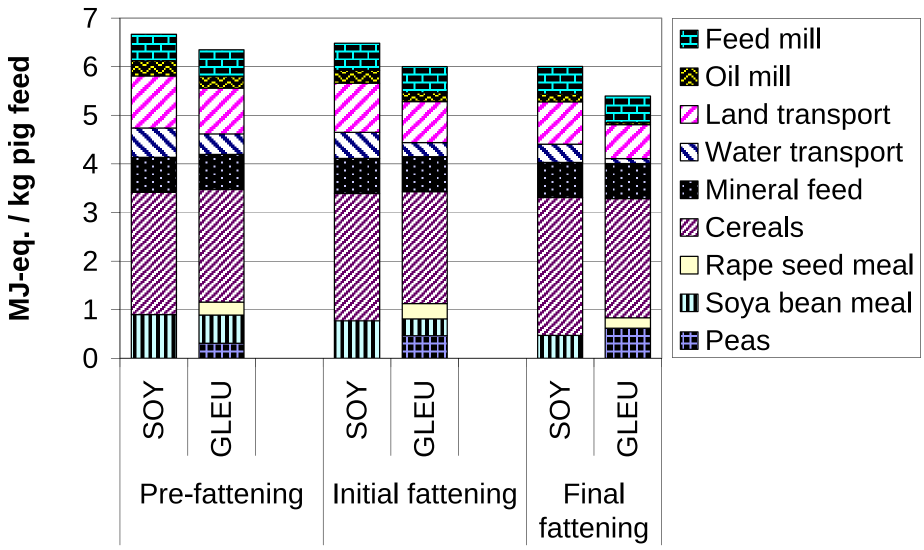


Fig. 1: Demand for non-renewable energy resources of six pig feed formulas in North-Rhine Westphalia shown in MJ-equivalents per kg pig feed. Pre-fattening: 20-40 kg; initial fattening: 40-65 kg; final fattening: 65-117 kg.

3.2 Eutrophication for chicken feed

For the impact category eutrophication the production of the raw materials is the dominant process, during which important losses of nitrate and phosphorus as well as the volatilisation of ammonia occur. The effects of the production of mineral feeds, transport and the processing at the oil and feed mill are of minor importance (Fig. 2). Comparing GLEU to SOY, the alternative containing European grain legumes is clearly less favourable than the one containing soya beans. The cultivation of the ingredients replacing soya beans, i.e. peas, oilseed rape and sunflowers, produces

higher nitrate leaching than of those in the protein part of the SOY formulas. This is due to the fact that soya has per weight unit a higher protein content than the replacing ingredients, which in return means that the share of the ingredients rich in protein is higher in the GLEU alternatives than in the SOY ones. Additionally, peas do supply a part of the energy requirements of the animal. Here pea is replacing maize in the GLEU formulas leading to a reduction of the eutrophication caused by the energy concentrates. This reduction though is far smaller than the increase of eutrophication due to the production of the protein concentrates (Fig. 2).
The results of the SAA alternatives are favourable compared to SOY. The main reason is the substitution of rape seed meal by maize gluten in the SAA formulas. The production of the latter causes a smaller impact on eutrophication.

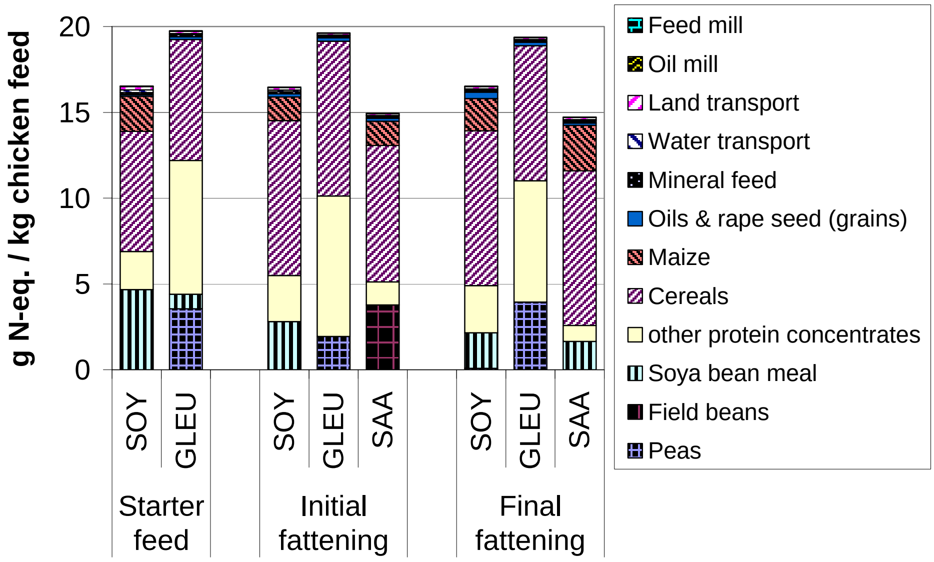


Fig. 2: Eutrophication for eight broiler (chicken) feed formulas in Brittany shown in g N-equivalents per kg chicken feed. Starter feed: 0-14 days; initial fattening 15-35 days; final fattening: 36-56 days.

3.3 Global warming potential for pig production

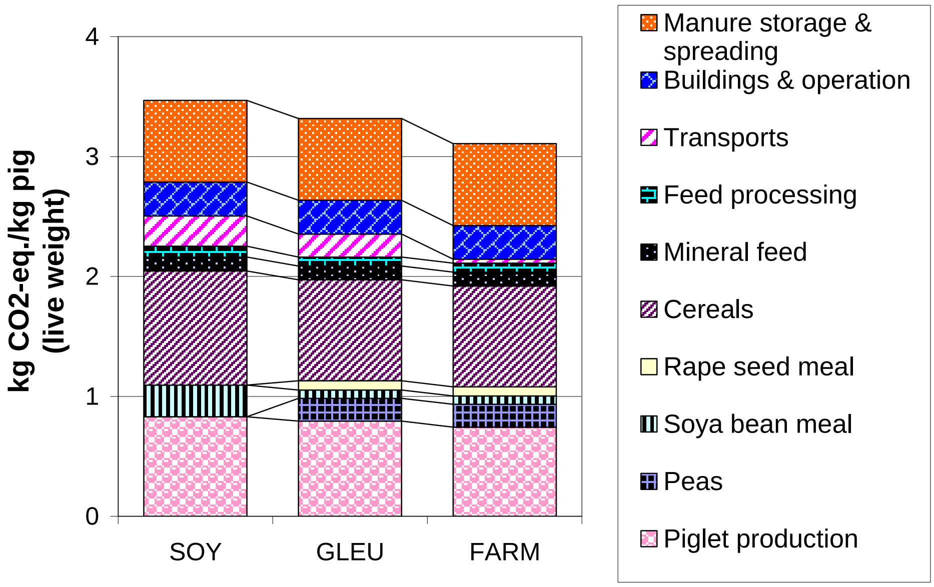


Fig. 3: Global warming potential (100 a) for pig production in North-Rhine Westphalia with three different feed alternatives shown in kg CO_2-equivalents per kg pig (live weight).
By enlarging the system assessed and taking the animal production (here pig) into account new process steps are added, as the piglet production, the building infrastructure and operation as well as

the manure storage and spreading. For the impact category global warming potential (GWP) we can observe that the production of the feed raw materials is still the part of the whole process chain with the greatest environmental impact. But piglet production – where feed production is included - as well as manure storage and spreading are also important contributors to this impact category, as is exemplified in figure 3.

Comparing the GLEU alternative with SOY there is a reduction of the GWP. Note that the reduction would be even higher, if additionally to the CO_2 emissions by fossil fuels the carbon release from deforestation would be considered. The reasons are less transports due to the substitution of soya beans from Brazil as well as a lower share of cereals in the GLEU formulas, as peas partly supplies the energy needs of the pigs. The reduction of the GWP from the alternative GLEU to FARM is even more important than from SOY to GLEU, which can be put down to the diminished transport (Fig. 3). This is a strong argument for local production of feed ingredients.

4. CONCLUSIONS

The replacement of South American soya beans by grain legumes (and other protein sources) produced in Europe reduces the environmental impacts in animal feed and meat production mainly for the impacts of resource management, e.g. energy demand or global warming potential. Whereas the impacts of grain legume use for nutrient (e.g. eutrophication) and pollutant management (e.g. terrestrial ecotoxicity, not shown in this paper) vary from favourable to unfavourable. Here the environmental impacts are clearly dependent on the choice of the accompanying protein concentrates replacing soya beans. The substitution of soya bean meal leads to a significant change in the feed composition. Using higher levels of synthetic amino acids seems to be a favourable option compared to formulas containing soya beans, but again the choice of the other ingredients can invert the results.

To moderate the environmental impacts from meat production, transport should be reduced by using local or even on-farm produced feed ingredients. The choice of the ingredients in the feed formulas is also of high importance. As crop production is the major factor, efforts in optimising environmentally the crop cultivation, e.g. better ratio between yield and inputs, should be undertaken. In equal measure, it should be aimed for optimising agricultural and nutritional crop properties, e.g. higher and stable yields or higher protein content of peas and field beans.

5. ACKNOWLEDGEMENTS

This research was supported by the European Commission (grant no. FOOD-CT-2004-506223) and by the Swiss State Secretariat for Education and Research.

6. REFERENCES

1. GLIP, 2004. The Grain Legumes Integrated Project: Project summary. http://www.eugrainlegumes.org/summary/index.htm
2. Crépon K., Cottrill B., Cechura L., Hucko J., Miguelañez R. & Pressenda F., 2005. Animal production sectors in seven European countries: Synthetic studies preceding the building of models simulating the raw materials supply of feed compounders. GLIP report. 80 p.
3. von Richthofen, J.S., Pahl, H., Nemecek, T., Odermatt, O., Charles, R., Casta, P., Sombrero, A., Lafarga, A., Dubois, G., 2006. Economic interest of grain legumes in European crop rotations. GL-Pro report, WP3.
4. Nemecek T., Heil A., Huguenin O., Meier S. Erzinger S., Blaser S., Dux D. & Zimmermann A., 2004. Life cycle inventories of agricultural production systems - ecoinvent data v1.1. Swiss Centre for Life Cycle Inventories, Dübendorf, CH.; ecoinvent report 15, 289 p.
5. Frischknecht, R., Jungbluth, N., Althaus, H.-J., Doka, G., Hellweg, S., Hischier, R., Nemecek, T., Rebitzer, G., Spielmann, M., 2004a. Overview and Methodology - ecoinvent data v1.1. Swiss Centre for Life Cycle Inventories (ecoinvent), Dübendorf; ecoinvent report 1, 75 p.
6. Nemecek T. & Baumgartner D., 2006. Environmental Impacts of Introducing Grain Legumes into European Crop Rotations and Pig Feed Formulas. Concerted Action GL-Pro, Final report WP4. ART, 63 S. http://www.art.admin.ch/themen/00617/00622/index.html?lang=de.

Environmental impacts of meat substitutes: comparison between Quorn and pork

S. Nonhebel and J. Raats
Center for Energy and Environmental Studies
Groningen University
Nijenborgh 4
9747 AG Groningen
The Netherlands
S.Nonhebel@rug.nl

Introduction

Meat substitutes are protein rich products, made from vegetarian ingredients resembling a piece of meat in shape, taste and texture. These products are not the same as vegetarian products, since that also includes products that don't resemble meat in any way, shape or form.
Meat substitutes are wanted by consumers since they want to include vegetarian ingredients in their diets, which are perceived to be healthier, to reduce weight, to be safer than normal meat products and they are perceived to be more environmental friendly (McIlveen, Abraham and Armstrong, 1999). Meat substitutes provide these vegetarian ingredients in a known form, so no big behavioral changes have to be made to take advantage of al these benefits. Also the fact that the price of meat substitutes is comparable to that of other meat products makes the difficulty of choosing meat substitutes over meat smaller. The market for meat substitutes is therefore not focused on "real" vegetarians, but more on "part-time" vegetarians, people that can do without meat in their meals every once in a while (McIlveen et al., 1999). Producing these meat substitutes from vegetarian products requires several production steps like heating, cooling, splitting, mixing, sieving etc. (Davies and Lightowler, 1998). Since these production steps all cause an extra environmental impact, the environmental impact of meat substitutes is higher than that of only vegetarian ingredients. But how big this environmental impact exactly is and how this environmental impact relates to that of meat is the main research question of this research
A meat substitute that is becoming very popular is Quorn, which is consisting out of fungi biomass, grown in large fermentators in the food industry.
In this paper we compare Quorn and pork with respect to their energy use, resource use and nitrogen emissions. Pork is with 50% of the total meat consumption the most common meat consumed in The Netherlands.

Production process Quorn

The raw material for Quorn in the fungus Fusarium venenatum A3/5. It is a filamentous fungus, which means that it forms long chains of cells (hyphae). The fugal structure composed by the hyphae is similar to the fibers of meat, which makes it possible to align the hyphae longitudinal and with that mimic the structure of meat. After this longitudinal alignment the similarities with meat are so big that the myco-protein has even been used as a reference standard for comparative tests on meat (Trinci, 1992). This resemblance of meat is important when looking at the acceptance of meat substitutes, as discussed in the introduction. Another advantage of using this specific fungus for biomass production is the fact that it can use various carbon sources as a substrate (Ugalde and Castrillo, 2002), like for example waste stream of the sugar industry called **molasses**.
The production process of Quorn starts in 2 bioreactors of each 155 m^3 in volume and 50 meters tall (Ugalde and Castrillo, 2002). These bioreactors are called external loop airlift fermentors (Christi, 1989). The process is a continuous flow process (Wiebe, 2001), which means the medium on which the fungus *F. venenatum* grows is pumped in at the same rate the culture is harvested from the fermentor (Trinci, 1992). The medium containing glucose as well as the, for growth necessary, nitrogen source and sterile air is added via an inlet at the base of the fermentors (Ugalde and Castrillo, 2002). In the fermentors these mixed raw materials, called a broth, are converted by the fungus into biomass by the fermentation process.
The oxygen pumped in is not only used for the growth of biomass, the oxygen bubbles also induce circulation and mixing of the broth caused by a difference in mean density between the riser and downcomer (Ugalde and Castrillo, 2002). At the top of the fermentor the pressure is reduced helping to release the waste product carbon dioxide (CO_2) produced by the fungus (Ugalde and Castrillo, 2002). Since the metabolic processes taking place within the fungus cause heat production, and the fungus grows optimally at 30 ºC, it is necessary to remove this heat (Solomons, 1985), using a water based cooling system (Coutouly, 2006) placed in the centre of the fermentor.
When the production process is operational about 300 kg of fungal biomass per hour per fermentor can be produced (Wiebe, 2001). With a biomass concentration of approximately 10 grams per liter medium this results in a total output of about 30.000 liters per hour per fermentor (Ugalde and Castrillo, 2002).
Since the produced fungal biomass has an RNA content of about 8-9% (Ugalde and Castrillo, 2002) the total output must quickly be heated to approximately 64 ºC to 65 ºC for about 20 to 30 minutes to reduce the RNA content to less

than 2%. This so called heat shock also reduces the amount of biomass output with 35% to 38% (Wiebe, 2001). After this heat shock the biomass is heated to approximately 90 ºC after which it is centrifuged to yield a paste containing approximately 30% solids (Trinci, 1992). This paste is then subsequently cooled to approximately 4 ºC. This paste is called myco-protein and is the raw material from which the Quorn products are produced.
The filaments in the myco-protein are first mechanically aligned after which a relatively small amount of egg albumen is added to stabilize the alignment. Also natural color and flavor ingredients are added followed by several heating, cooling and formation steps. Finally the product, now called Quorn, is reduced in size and put into cold storage for further packaging and distribution.

Production process pork

In the Netherlands the production of pigs mainly takes place by so-called intensive farming, the necessary feed for the pigs is in this case not produced on the farm itself but imported from other sites (Gerbens-Leenes, 1999). The first part of the production of pigs takes place in the so-called farm stage. In the Netherlands the pigs are held in barns during this farm stage. These barns on average have floor heating, extra heating from heat lamps for the piglets (young pigs) and are also ventilated and lighted. In the livestock production sector a large part of the total energy is used for these kinds of processes (Elferink, Nonhebel and Moll, 2006).
To feed the pigs during this farm stage normally an optimal feeding plan (Technisch Model Varkensvoeding, TMV) is used. This feeding plan calculates an optimal feed for pigs using a technical model (Centraal Veevoeder Bureau (CVB), 2003), using a combination of several different feeds and taking into account the amount of nutrients, like the before mentioned carbon and nitrogen, needed by the pigs. The organic waste stream of the sugar industry, for a significant part **molasses**, is an example of an important supplier of raw materials for the fodder industry (Elferink, 2001).
The real production of pigs in the farm stage starts with the fertilization of the sow (female pigs). After fertilization it takes on average approximately 115 days before the piglets are born. On average approximately eleven piglets are born per litter. After birth the piglets stay with their mother for weaning for about four to five weeks after which they are separated from each other. After this so-called nurturing stage, when the piglets are about ten weeks old, they are sent to a finishing barn. In this finishing barn the piglets are kept for six to eight months When the live weight of the pigs is approximately 110 kg (assuming the average carcass weight (meat with bone) of approximately 90 kg (PVE, 2006) is 81% of the live weight of a pig (Elferink and Nonhebel, 2006)) the pigs are further slaughtered and processed. This is the so-called slaughtering and processing stage.
According to Ramirez Ramirez (2005) this further slaughtering and processing stage contains 10 different steps. First of all the pigs are stunned and slaughtered. After this first step the blood is removed from the pigs and the pigs are scalded at about 58 °C to 65 °C. After scalding the pigs are singed at approximately 900 °C to 1000 °C after which they are trimmed. Then the intestines are removed and the whole pig is splitted in two. When this is finished the pork is chilled to approximately 7 °C and cut and boned
Result is that approximately only 56% of the live weight of a pig is ready for consumption by humans (PVE, 2006), which equals 69% of the carcass weight of a pig. This fraction is further referred to as the human edible portion (HEP). In a table these relative portions look as follows (Table 3.1).

System boundaries

Since both products could potentially grow on the raw material molasses the amount of this raw material needed for production was chosen to act as an environmental indicator. Also the amount of nitrogen needed for the production of both products was chosen as an environmental indicator. Nitrogen is used in both products to produce proteins, which are necessary for constructing and maintaining the human body. Since molasses (the main carbon source) and nitrogen are the two major macronutrients used by most organisms in the highest abundance (Madigan et al., 2000), they are therefore probably the materials that cause a relatively high share of environmental impact. Besides molasses and nitrogen, the amount of energy needed to produce these products was also chosen to act as an environmental indicator. Energy use is chosen because many environmental problems are related to energy use (Duthil and Kramer, 2000) and the production of both products is assumed to consume energy.
The system boundaries of this research therefore start on the production facilities of Quorn and pork themselves. It is assumed for both processes the molasses can be used for the process without need of a pre-treatment. The subsequent production stage and its indirect effects are said to be important when discussing environmental impacts of the total product (Jungbluth, Tietje and Scholz, 2000). For both products the amounts of raw materials together with the amount of energy needed to convert these raw materials into Quorn or pork are calculated.

Results and conclusion

Table 1: Amounts of resources required to produce 1 kg human edible pork or 1 kg of Quorn

	Pork	Quorn	Ratio (Pork/Quorn)
Molasses required	8,3 kg	2,9 kg	2,9
Nitrogen required	132 gram	69 gram	1,9
Energy required	12,1 MJ	13,6 MJ	0,9

With respect to resource use and nitrogen emissions the Quorn system proved to be better, but with respect to energy the pork production system gave better results. This implies that not on all environmental impacts this meat substitute (Quorn) is better for the environment than pork.

Acknowledgement

This paper is based on: Raats, J. 2007. Meat(substitutes) comparing environmental impacts. A case study comparing Quorn and pork. Master thesis, Center for energy and environmental studies, Groningen University, Groningen, The Netherlands.

References.

Centraal Veevoederbureau (CVB). (2003). *Tabellenboek Veevoeding 2003.* Lelystad.

Cooney, C., Wang, D.I.C., and Mateles, R.I. (1969). Measurement of heat evolution and correlation with oxygen consumption during microbial growth. *Biotechnology. Bioengineering, 11*, 269-281.

Christi, M.Y. (1989). *Airlift Bioreactors*. New York: Elsevier Science Publishers

Davies, J., and Lightowler, H. (1998). Plant-based alternatives to meat. *Nutrition and Food Science*, *2*, 90-94

Dutilh, C.E., and Kramer, K.J. (2000). Energy consumption in the food chain: comparing alternative options in food production and consumption. *Ambio, 29* (2), 98–101

Elferink, E.V. (2001). Vlees*: een duurzame eiwitbron?: Een modelstudie naar het inzetten van organische reststromen in de veehouderij*. Master-thesis. Rijksuniversiteit Groningen, Groningen.

Elferink, E.V., and Nonhebel, S. Variations in land requirements for meat production. *Journal of Cleaner Production, In press.*

Elferink, E.V., Nonhebel, S., and Moll, H.C. (2007). *Variation in energy use of livestock food products*. Submitted to Energy Policy

Gerbens-Leenes, P.W. (1999). *Indirect Ruimte- en energiebeslag van de Nederlandse voedselconsumptie*. Ivem-research report, 102. Rijksuniversiteit Groningen, Groningen

Hoek, A.C., Luning, P.A., Stafleu, A., and de Graaf, C. (2004). Food-related lifestyle and health attitudes of Dutch vegetarians, non-vegetarian consumers of meat substitutes, and meat consumers. *Apetite, 42*, 265-272.

Jungbluth, N., Tietje, O. and 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* (3), 134–14

Madigan, M.T., Martinko, J.M., and Parker, J. (2000). *Brock Biology of Microorganisms: Ninth Edition*. Upper Saddle River, NJ: Prentice-Hall.

McIlveen, H., Abraham, C., and Armstrong, G. (1999). Meat avoidance and the role of replacers. *Nutrition and Food Science*, *1*, 29-36

Productschappen Vee Vlees en Eieren (PVE)., Voorlichtingsbureau Vlees. (2004). *Vlees, cijfers en trends 2003: Marktverkenning over het consumptiegedrag in een dynamische samenleving*. Zoetermeer

Ramirez Ramirez, C.A. (2005). *Monitering energy efficieny in the food industry*. Phd-thesis. Universiteit Utrecht, Utrecht.

Trinci, A.P.J. (1992). Mycoprotein: a twenty year overnight success story. *Mycological Research, 96*, 1–13.

Ugalde, O.U., and Castrillo, J.I. (2002). *Single Cell Proteins from Fungi and Yeasts*. Retreived December 8, 2005, from http://www.sc.ehu.es/qpwugmau/principal/RevUgalde.PDF

Wiebe, M.G. (2004). Quorn^{tm} Myco-protein: Overview of a successful fungal product. *Mycologist, 18* (1), 17-20.

Zhu, X., and Van Ierland, E. (2004). Protein chains and environmental pressure: A comparison of pork and novel protein foods. *Environmental sciences*, 1 (3), 254-276.

Comparing the energy use of different livestock product systems

Elferink, E.V., S. Nonhebel.
Center for Energy and Environmental Studies (IVEM)
University of Groningen
Nijenborgh 4
9747AG Groningen
The Netherlands
Tel: (++)31-50-3634611, fax (++)31-50-3637168
E-mail: e.v.elferink@rug.nl

Introduction

The production of food requires 20% of total energy use. Therefore, energy efficiency improvements in food production and consumption can potentially contribute considerable to fossil energy reduction. Animal based food products are especially regarded as energy intensive (Carlsson-Kanyama et al. 2003; Kramer and Moll 1995). An often mentioned possibility to reduce the energy use of food production and consumption is, therefore, a change to a more vegetarian diet (Carlsson-Kanyama 1998; Goodland 1997; Zhu and van Ierland 2004). However, consumers prove to be very reluctant to discard the consumption of livestock food products (Nonhebel and Moll 2001). In the short-term consumption changes to a vegetarian diet are, therefore, not likely to occur. To reduce the energy use of livestock food production in the near future other options need to be explored as well.

Food products of animal origin as meat, milk and eggs satisfy for a large part the protein requirements of humans. Livestock food products are, however, produced in quite different production systems by different livestock. Livestock differs, for instance, in feed requirements but also in the way they are kept and how their products are processed, stored and prepared.
Differences between production sectors of livestock food products will result in different energy uses. However, presently an overview is lacking. Insight in the energy use of different livestock food production systems will increase the understanding on how to improve the energy efficiency of livestock food products. This study focuses on the energy use in the livestock food production system of chicken, pork, eggs and milk in the Netherlands.

Method

An energy system analysis is used to determine the energy use of the Dutch conventional livestock production system. Livestock food products are produced in a food supply system that exists of multiple production sectors. The analysis includes all phases of the production system. A bottom-up inventory is executed for each of the livestock food products studied showing the energy used in each phase. The descriptions of the livestock food production systems needed for such an inventory are based on literature and expert interviews. Data on energy use are acquired from literature, statistical yearbooks, online databases and specialist journals. Figure 1 shows the simplified livestock food production system as analyzed in this study.
Livestock food products are a part of the human diet providing essential micro- and macronutrients but most of all livestock food products are a source of high quality dietary proteins. Therefore, data in this paper is not only presented on a kilogram product base but also on the bases of the protein content.

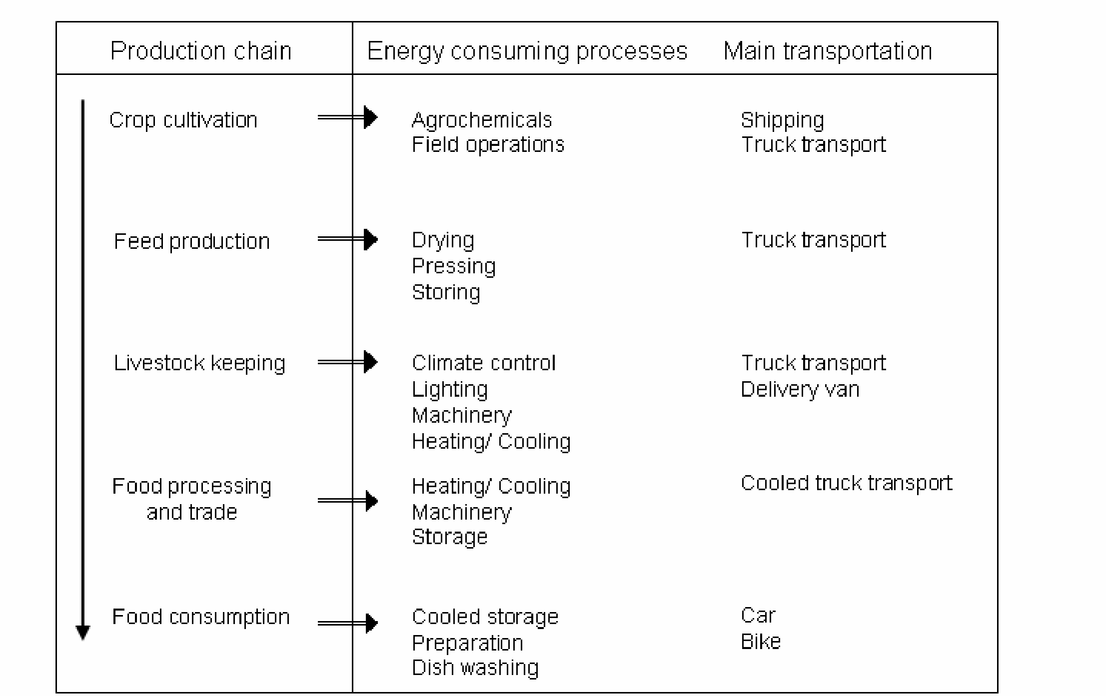


Figure 1 The production system of livestock food products, the related main energy consuming processes and transportation means.

Results

The energy used to produce and consume livestock food products are shown in figures 2 and 3. When compared on fresh weight the spread in energy use between the livestock food products studied is large (figure 2). However, when compared on protein content the spread in energy use is reduced significantly (figure 3). For instance, chicken have with 78 MJ/ kg product the highest energy use while milk has with 9 MJ/ kg product the lowest energy use. On protein content chicken have again the highest energy use with 388 MJ/kg protein. Eggs have, however, with 226 MJ/kg protein the lowest energy use

Analysing the energy use of livestock food products

Pork has an energy use of 350 MJ/kg protein the share of each production step to the total energy use differs, however, substantial e.g. transport is with 130 MJ/kg protein the largest energy consuming sector while food processing has with 20 MJ/kg protein the lowest energy use. For the production of chicken 390 MJ/kg protein is needed. The largest energy consuming sector within the chicken system is, however, the livestock sector with 140 MJ/kg protein while food processing has with 32 MJ/kg protein the lowest energy consumption. Eggs require a total energy use of 227 MJ/kg protein. Transportation and food consumption are the largest energy consuming sectors with respectively 80 MJ/kg protein and 78 MJ/kg protein. Food processing requires only 2 MJ/kg protein and is the least energy consuming sector. The energy use of milk is 250 MJ/kg protein. The livestock sector has with 10 MJ/kg protein the lowest energy consumption while the production of feed requires the highest amount of energy, 84 MJ/kg.

Evaluating the differences in energy use per sector

There are large differences in energy use within the same production sector among livestock food products.

The energy required for feed production depends on the feed conversion factor, the feed ingredients used, the energy required to produce a specific feed (roughage feed or concentrate feed) and the share of a feed in the diet. Livestock has a high feed conversion factor compared to the other livestock. As a result feed production is for milk the highest energy requiring sector, 84 MJ/kg protein.

The large difference in energy requirements in the livestock sector is mainly due to the heating of broiler chickens with heating/breeding lamps ,140 MJ/kg protein. In contrast dairy cows are kept partly outdoors or in stables which are not heated at all 10 MJ/kg.

Eggs need hardly any handling or processing and are not cooled at the retailer as a result eggs have the lowest energy requirement for food processing, 2 MJ/kg protein. In contrast milk has an high energy requirement, 33 MJ/ kg protein, due to the high water content of milk combined with some energy intensive production steps for processing whole milk into pasteurized milk.

Transportation of feed ingredients comprises the majority of the energy used within the transportation sector, depending on the livestock type between 68% to 82%. Feed for pork exists largely out of feed ingredients that are cultivated in countries outside the Netherlands or EU (e.g. soybean meal, tapioca). As a result, pork uses 130 MJ/kg protein for transportation. Although dairy cattle have a high feed conversion factor the relative low energy requirement for transport, 61 MJ/kg protein, is because dairy cattle are fed with feed that is largely produced in the Netherlands.

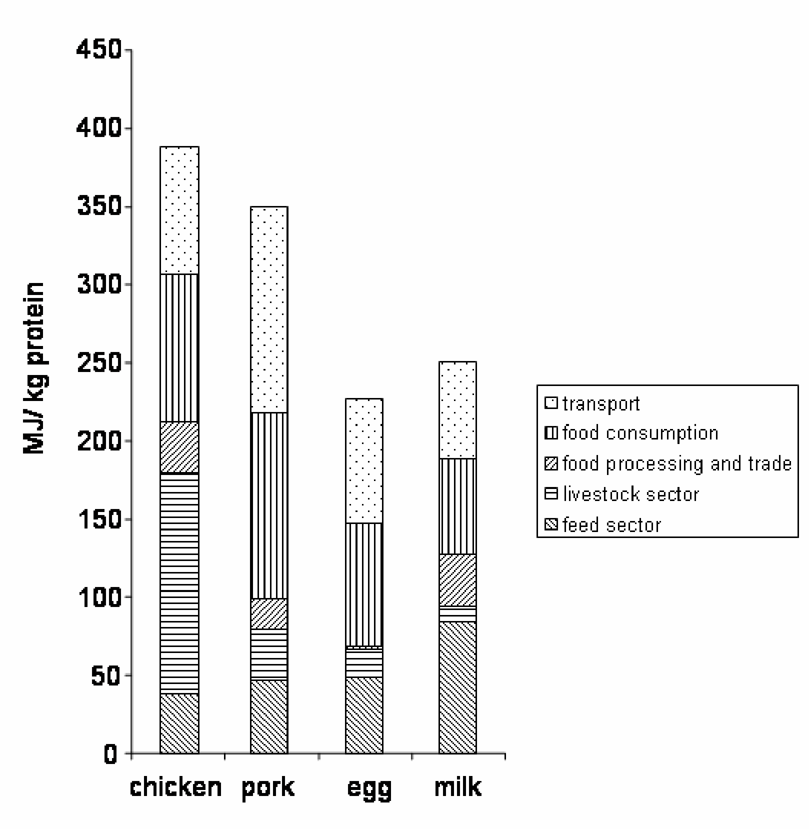


Figure 2 Distribution of energy use in the production system of livestock food products produced in the Netherlands.

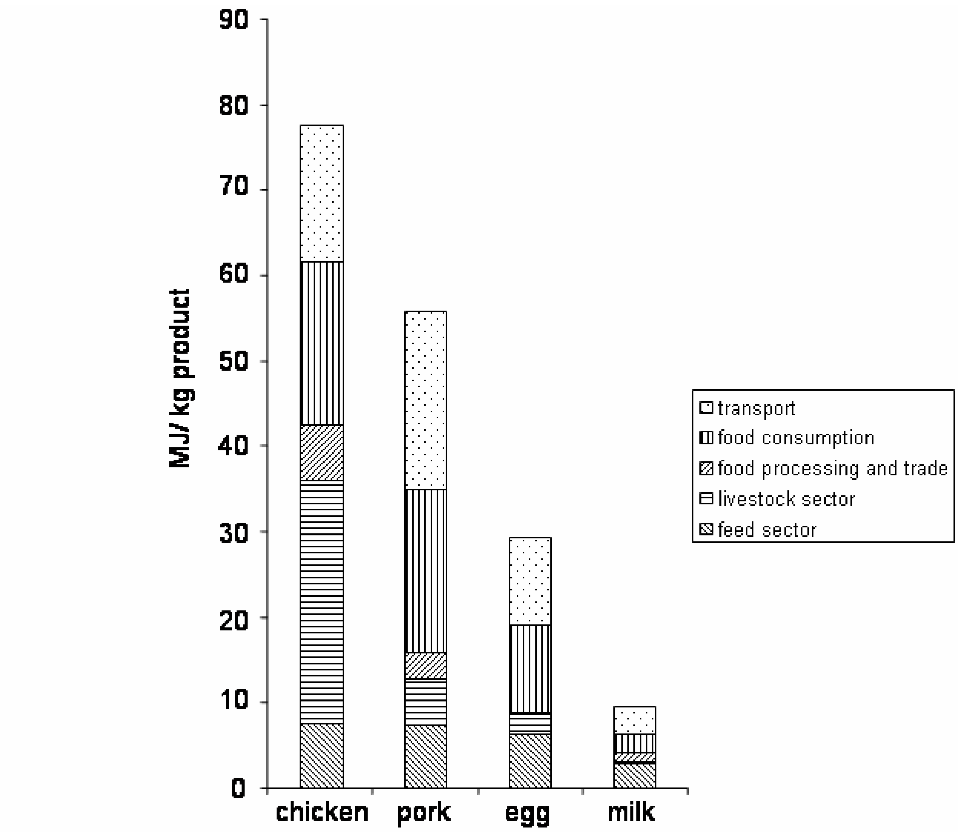


Figure 3 Distribution of energy use in the production system of livestock food products produced in the Netherlands represented per sector

Conclusion

The system analysis showed large differences in energy use among livestock food products and the different sectors of the livestock production system studied. When compared on a protein base the energy use of livestock food products differ by a factor 1.7. In the Netherlands is eating eggs instead of milk instead of pork instead of chicken more energy efficient. The large differences between production sectors of a livestock food product show a large potential for energy reduction options.

References

Carlsson-Kanyama, A. 1998. "Climate change and dietary choices -- how can emissions of greenhouse gases from food consumption be reduced?" *Food Policy*. 23:277-293.

Carlsson-Kanyama, A., M.P. Ekstrom, and H. Shanahan. 2003. "Food and life cycle energy inputs: consequences of diet and ways to increase efficiency." *Ecological Economics*. 44:293-307.

Goodland,R. 1997. "Environmental sustainability in agriculture: diet matters." *Ecological Economics*. 23:189-200.

Kramer, K.J. and H.C.Moll. Energie voedt. 77. 1995. Center for energy and environmental studies (IVEM).

Nonhebel, S. and H.C. Moll. Evaluation of Options for Reduction of Greenhouse Gas Emissions by Changes in Household Consumption Patterns. 2001. Groningen, University of Groningen. Dutch National Research Programme on Global Air Pollution and Climate Change.

Statistics Netherlands (CBS). Statline. 2005.

Zhu,X. and E. van Ierland. 2004. "Protein Chains and Environmental Pressures: A Comparison of Pork and Novel Protein Foods." *Environmental Sciences*. 1:254-276.

Development of a Sustainability Indicator for Agro-Food Consumption and Production

Toshisuke OZAWA[a] , Kiyotaka TAHARA and Atsushi INABA[a,b]

[a]Research Center for Life Cycle Assessment
National Institute of Advanced Industrial Science and Technology (AIST)
16-1 Onogawa, Tsukuba, Ibaraki 305-8569 JAPAN
[b]Research into Artifacts, Center for Engineering (RACE), The University of Tokyo
Kashiwanoha 5-1-5, Kashiwa, Chiba 277-8568, JAPAN

Abstract

The Food Study Group (FSG), a voluntary-based study group in the Institute of Life Cycle Assessment Japan, carries out research to identify the possible directions to sustainable food consumption and production. FSG had mainly two tasks: (1) life cycle inventory analysis on food products and meals; and (2) determination of food values to develop a sustainability indicator for agro-food consumption and production. For the latter task, FSG adopted the concept of eco-efficiency, that is evaluated by comparing a concerning product's service value with its environmental loads. For product value, we discussed the definition, criteria, and the method to quantify the value of a meal. The candidates for criteria for this study are: variety in food group intake and nutrients intake from three meals a day. As a first attempt in such a novel study, we quantified the values of the model meals and eco-efficiency with our method.

1. INTRODUCTION

Carbon dioxide (CO_2) emission induced from food consumption is the third highest after utility (gas, electricity, water) and transportation from an average household in Japan among household living expense items. At a global level, measures and strategies for environmentally sound food consumption and production is becoming high in the agenda of sustainability. In order to turn the on-going agro-food consumption and production pattern into a more sustainable mode, quantification of environmental load on concerning products throughout their entire lifecycle is a prerequisite. Moreover, it is necessary to seek acceptable measures in consumption and production patterns that could increase or maintain consumers' quality of life (such as health and convenience in food). In this paper, the concept of eco-efficiency (WBCSD, 1992) was adopted for the development of a sustainability indicator. Eco-efficiency is evaluated by comparing a concerning product's service value with its environmental loads. Life cycle assessment (LCA) is used to estimate CO_2 emissions to quantify environmental burden from food items and meals. However, to our knowledge, there is no report on quantification of food value, as it has been considered as one of the biggest obstacles because the value depends heavily on human' subjectivity and diversity in liking. The purpose of this study is to make a first attempt to quantify food value by scoring the concerning meals based on consumers' objective criteria instead of subjective criteria.

2. THE SUGGESTED INDICATOR

2.1 . Concept of the indicator

The following is the suggested eco-efficiency indicator for this study:

$$\textit{Eco-efficiency} = \frac{\textit{Value that a consumer receives from having meals in a day}}{\textit{LC-CO}_2\textit{ from meals served in a day}} \quad \text{....(1)}$$

In this study, we calculated the value that a consumer receives from having meals in a day. Evaluation scope for environmental burden was set equivalent for the food vale and was LC-CO_2 from meals served in a day (a total of breakfast, lunch and dinner). Tsujimoto and Tsuda (2006) set forth model menus of breakfast, lunch and dinner (Japanese, Western and Chinese dish) and calculated CO_2 emissions from house-cooked meals by combining the CO_2 emission of the ingredients and that of direct energy consumption through cooking. Their results were adopted for the environmental burden for eco-efficiency. The food value from equation 1 is a total of the values that consumer receives from having breakfast, lunch and dinner in a day.

Next, the value that consumer receives from having a meal (V_{meal}) is expressed in equation 2.

$$V_{meal} = \frac{T_i}{T_{\max}} * W_T + \frac{N_i}{N_{\max}} * W_N + \frac{H_i}{H_{\max}} * W_H \qquad \text{......(2)}$$

where T=taste, N=nutrient balance, H=health function (non-subjective criteria) and W=weighting of importance (determined via consumer survey). The procedure of quantification of the food value are: (1) quantify the rating of each criterion (taste, nutrient balance, health, etc.); (2) adopt relative evaluation to each criterion among the food items compared, to normalize the rating; and (3) if necessary, multiply the normalized scores by weighting factors to derive the values. The calculated values are object-specific; therefore, the criteria can be varied by users depending on the objects evaluated.

2.2．Selection of criteria and weighting

We decided that the food value be calculated based on criteria of consumer requirements toward food. In a consumer survey conducted by the Energy Conservation Center, Japan (ECCJ 2005), there are questions about the important requirements in food life. We categorized the requirements and summarized in Table 1.

Table 1 Categorized criteria for food valuation from ECCJ consumer survey (2005)

a. Objective (external) category	b. Subjective (internal) category	c. Other
• Sufficient Nutrients	• Tasty	• Economical (price)
• Variety in kinds of ingredients	• Enough time to enjoy meal	• Easy cooking
• Products of the home area	• Access to anything desired	• Safe
• Products of its season	• Fresh from oven	• Good for health
• House grown vegetable	• Conversation with family	

We selected the requirements particularly important for consumers and used them as criteria for valuation of meals. The requirements were categorized into objective (eternal), subjective (internal) and other requirement items. It is speculated that since food is closely attached to emotion and subjectivity, they occupy the major decision-making factor. Therefore, we should use subjective criteria for the determination of food value. However we adopted objective criteria for that purpose because it is difficult to quantify subjectivity with engineering means. Also, we did not adopt any of the “other” items this time for the following reasons: safety is a prerequisite for food, easy cooking is a requirement mainly from cookers. Economical can be treated as a holistic criterion with monetary figure that includes all the subjective and objective evaluation criteria. In estimating weighting factor, one must consider the variety in respondents’ attributes, attitude toward food at different life scenes, life style, etc. However, we simply used the mean value for each criterion from the questionnaire survey with 3,664 respondents. Details on these aspects will be discussed in the future study.

3. CASE STUDY

3.1. Determination of food values

We performed a case study aiming at making the suggested eco-efficiency into a usable indicator in food consumption and production. For this case study, we chose food groups sufficiency level (18 items from 4 groups) and nutrient intake (25 nutrients, energy, salt content and vegetable intake - 28 items) as criteria for food value. These criteria are both objective category and readily be quantified. We referred to the Health, Labor and Welfare Ministry's (HLWM's) "National Survey Report of Heal and Nutrient Situation in Japan" (2003) for the details on the science based dietary intake reference value (for one day). For quantification of food value, we set the calculation condition as the following:

- Values for risk of insufficiency and risk of health hazard from excess intake are given for some nutrients in HLWM's report. When both reference values are given, the score is 100 as long as the intake value falls in that range.
- When insufficient (reference value > intake), Value = Intake/Reference*100.
- When excess intake (reference value < intake), Value = Reference/Intake*100.
- If only one value is given, the score reaches 100 as intake value approaches to reference value.

There are some nutrients that are missing the upper limits. This does not mean that there is no upper limit for intake, but rather mean that enough scientific evidence does not exist.

3.2. Determination of environmental burden

CO_2 emissions from house-cooked meals were calculated by combining the CO_2 emission of the ingredients and that of direct energy consumption through cooking. The summary of LCA results on model meals is shown in Table 2. The LC-CO_2 including ingredients and cooking with I-O data and process LCA for breakfast, lunch, dinner 1, 2 and 3 are 1,172, 1,940, 3,084, 5,855 and 2,958, respectively. The details on the determination of environmental burden are described in Tsujimoto and Tsuda (2006) in Japanese and Ozawa and Inaba (2007) in English.

4. RESULTS AND DISCUSSION

The food values determined by food groups sufficiency level and by nutrient intake with using our suggested quantification method are shown in Figure 1 and Table 2. There were no large differences in food groups sufficiency level or nutrient intake among the model menus as expected. The menus and applicable amounts of food items per serving for Japanese adults to stay healthy were determined based on the Health, Labor and Welfare Ministry's Japanese Food Guide Spinning Top (2006). Therefore, the amounts of nutrients taken through the model menu are supposed to be sufficient.

The calculated eco-efficiency after aggregation is shown in Table 2. The weighting of evaluation criteria were determined by consumer survey. The weightings for sufficient nutrient and variety in ingredients were 71.9 and 9.5 out of 100, respectively. These weightings were multiplied by values for nutrient intake and food group sufficiency level, respectively, so that the aggregated food values for menu 1, 2 and 3 are 0.61, 0.62 and 0.61, respectively. These values were divided by environmental burden (kg-CO_2/day) of 6.20, 8.97 and 6.07 for menu 1, 2 and 3, respectively to calculate eco-efficiency. The aggregated values for all the menus are close to each other, however, eco-efficiency for menu 2 resulted in the lowest among them because menu 2 has larger environmental burden than the others by 2.7 kg-CO_2/day (30%). Currently, we are working on defining the scope of application of suggested eco-efficiency indicator.

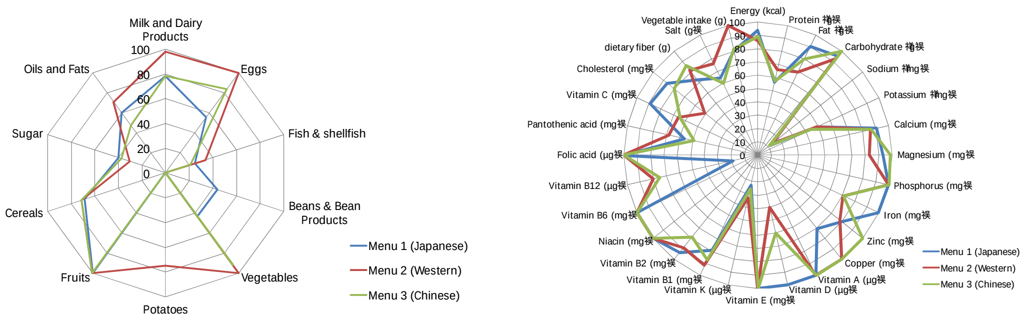


Figure 1 Raider charts of the food values determined by food groups sufficiency level (left) and by nutrient intake (right)

Table 2 Eco-efficiency calculated by the values calculated with the suggested method

Weighting of evaluation criteria from survey results

	Frequency	Weighting
Sufficient Nutrients	2,634	71.9
Variety in ingredients	349	9.5
Products of its season	423	11.5
House grown vegetable	89	2.4
Products of home area	48	1.3
Other	121	3.3
Total	3,664	100.0

	Value		
	Food group sufficiency level	Nutrient intake	Aggregation
Menu 1	0.51	0.78	0.61
Menu 2	0.68	0.77	0.62
Menu 3	0.53	0.78	0.61

	Environmental Burden ($g-CO_2/day$)				
	BF	Lunch	Dinner	Total	$kg-CO_2/day$
Menu 1	1,172	1,940	3,084	6,196	6.20
Menu 2	1,172	1,940	5,855	8,967	8.97
Menu 3	1,172	1,940	2,958	6,070	6.07

Eco-efficiency	
Menu 1	0.13
Menu 2	0.09
Menu 3	0.13

5. CONCLUSIONS

- We made an attempt to calculate eco-efficiencies of three different model meals based on the quantification method of food value using food group sufficiency level and nutrient intake as evaluation criteria.
- All three food values are almost the same because each menu was determined based on sufficient intake of nutrient and food groups.
- Scope of application of suggested eco-efficiency indicator will be defined soon.

6. REFERENCES

1) World Business Council for Sustainable Development (WBCSD), (1992), “Changing Course,” the report to Rio Summit.
2) Susumu Tsujimoto and Toshie Tsuda (2006), LC-CO2 evaluation of house-made food by setting the model menu, 2nd Conference of the Institute of Life Cycle Assessment, Japan, P2-10.
3) Energy Conservation Center, Japan (2005), Report on the direct and indirect energy consumption survey in food life.
4) Health, Labor and Welfare Ministry’s “Japanese Food Guide Spinning Top” (2006)
5) Health, Labor and Welfare Ministry, (2003) National Survey Report of Heal and Nutrient Situation in Japan.
6) Toshisuke OZAWA and Atsushi INABA, (2007), *“Introduction of the Food Study Group, the Institute of Life Cycle Assessment, Japan,”* in the same proceedings.

Exploring a comparative advantage for New Zealand cheese in terms of environmental performance

Claudine Basset-Mens[1], Sarah McLaren[1,2], Stewart Ledgard[1]

1. AgResearch Limited, Ruakura Research Centre, East Street, Private Bag 3123, Hamilton, New Zealand, Tel. +64 7 838 5213, Email. claudine.basset@agresearch.co.nz
2. Landcare Research, PO Box 40, Lincoln, New Zealand.

Abstract

It has become increasingly important to assess the environmental performance of the complete life cycle of NZ food products, as the debate about "food miles" is intensifying. A complete life cycle assessment of NZ cheese has been undertaken and the contribution of transportation stages estimated for five impact categories. The farm stage was the main contributor to all impact categories (39-87%). All transportation stages together represented a hot spot only for energy use (38% of total life cycle energy use); however, shipping itself only represented a minor part of total energy use. Sewage treatment was found to be important for eutrophication and energy use for dairy manufacturing. A sensitivity analysis indicated that NZ cheese has some advantages compared with EU cheeses in terms of environmental performance even when accounting for shipping. This needs to be validated with a comparison between UK/EU systems and the NZ system over their cradle to grave life cycle using harmonised methodology.

1 INTRODUCTION

Most NZ food products are exported worldwide and have to be transported over long distances to reach their markets. The "food miles" concept (defined as the distance food travels from producer to consumer, [1]) has become popular in the UK and USA and potentially threatens market access of NZ food products to distant markets. The purpose of this study was to analyse the relevance (or not) of "food miles" in the debate over sustainability and export of NZ food products. Dairy products were selected because of the importance of the dairy industry to the NZ economy. In 2004, 660,000 tonnes of wholemilk powder and 296,000 tonnes of cheese were exported from NZ [2]. Dairy products comprise 17% by financial value of all exports from NZ [3]. Cheese exported to the UK was selected as a typical situation potentially under threat: the UK is one of the furthest export destinations for NZ products and there is a high level of awareness about "food miles" in this country. A life cycle assessment from "cradle to grave" was performed for NZ cheese and the contribution of transportation stages was estimated. The comparative advantage of NZ cheese in terms of environmental performance compared to its locally produced equivalent was also explored.

2 MATERIALS AND METHODS

2.1 Dairy product life cycle

The entire life cycle of cheese was studied, from production of fertilisers, electricity and capital for the agricultural stage through to sewage treatment after consumption of the products. An average NZ farm for the year 2005 was used [4]. Cheese was assumed to be processed in the major dairying region of NZ (Waikato) and distributed to the UK by shipping (Fig. 1).

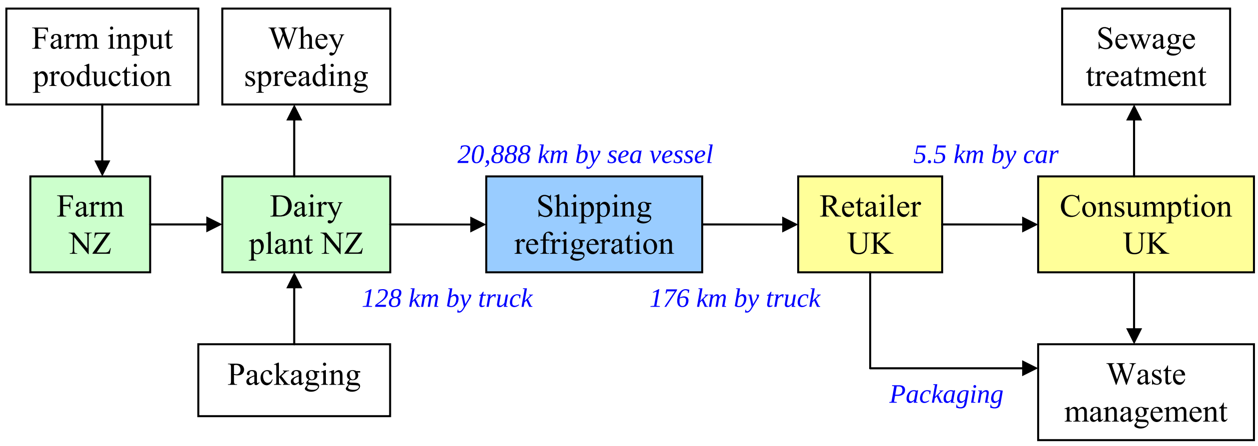


Figure 1. Life cycle of cheese in the study (distances are the one-way distances)

2.2 Inventory data

Inventory data for the "cradle to farm gate" analysis have been presented in [4]. Only old data for energy used in dairy manufacture were available [5] but they were similar to more recent data from Denmark [6]. Cheese was assumed to be refrigerated for all transportation stages and during storage at the retail distribution centre and at home. For sewage treatment, the method presented by [7] was followed. Specific industry data were used for transportation from farm to factories and from factories to NZ port. Transportation distance by ship was according to [8] and for onward transport to retailer and on to the home data were taken from [9].

2.3 Impact assessment

The "CML 2 baseline 2000" method was used in this study. The impact categories selected were climate change (100 years), acidification and eutrophication. Land use (only to the farm gate) and energy use (cumulative energy demand) were also calculated.

2.4 Sensitivity analysis to product origin

In order to explore a possible comparative advantage for NZ cheese, a sensitivity analysis was conducted to the product origin using cradle to farm gate LCA studies from Sweden [10], The Netherlands [11] and from the UK [12]. For these EU systems, the specific cradle to farm gate data were used, the shipping stage was omitted, and all downstream stages were assumed to be the same as for the NZ cheese life cycle, as a first approximation.

3 RESULTS AND DISCUSSION

3.1 Environmental performance and hot spots of NZ cheese life cycle

The environmental impacts of NZ cheese are presented in Table 1. For all impact categories, farm production had the greatest contribution, ranging from 39% (energy) to 87% (acidification) (Fig. 2). Dairy manufacturing had a significant contribution only for energy use at 18% of the total energy use. Sewage treatment was the second main contributor for eutrophication after farm production at 31%. All transportation stages (truck, ship, car) contributed from 2.5% (eutrophication) to 38% (energy). Except for acidification, the contribution of ship transport to each impact was less than the contribution of truck and car transport together. For energy use and GWP, ship transport represented 27% of all transportation stage impacts. Overall, ship transport contributed only 1% (eutrophication) to 10% (energy) of each impact.

Table 1. Environmental impact assessment of NZ cheese per kg of product and sensitivity analysis to cheese origin, using Swedish, Dutch and UK "cradle to farm gate" data (difference as a percentage of NZ data are given in parentheses)

Per kg of product	GWP (kg CO_2-eq)	Acidification (kg SO_2-eq)	Eutrophication (kg PO_4-eq)	Land use (m^2.a)	Energy use (MJ)
NZ	10.0	0.085	0.044	10.2	39.1
Sweden[1]	11.9 *(+18%)*	0.180 *(+111%)*	0.075 *(+71%)*	18.5 *(+82%)*	54.2 *(+39%)*
Netherlands[2]	14.9 *(+48%)*	0.098 *(+15%)*	0.151 *(+247%)*	12.3 *(+21%)*	68.2 *(+74%)*
UK[3]	11.2 *(+12%)*	0.158 *(+86%)*	0.076 *(+75%)*	10.2 *(+0%)*	43.1 *(+10%)*

1: [10]; 2: [11]; 3: [12]

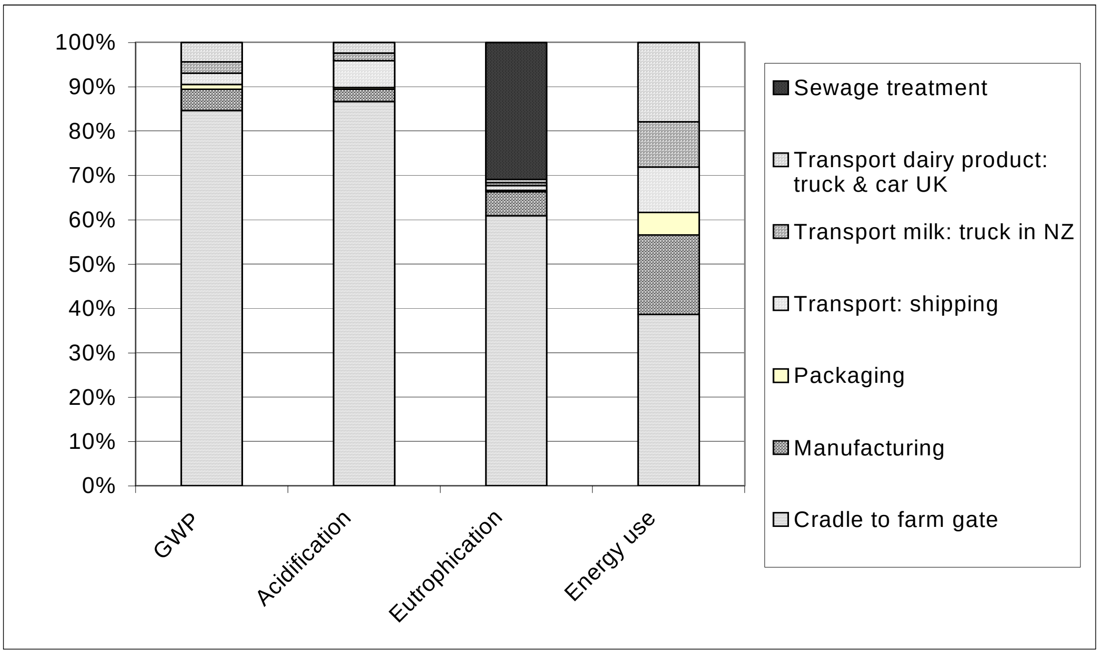


Figure 2. Contribution analysis of GWP, Acidification, Eutrophication and energy use for NZ cheese

3.2 Sensitivity analysis to product origin

In our sensitivity analysis of the cradle to grave impacts of cheese, NZ cheese had similar or less impact than EU systems (Table 1). Big variability across EU studies can be noted, probably due both to country specificities (climate, practice) and differences in methodologies. In comparison with the UK system, NZ cheese was similar in terms of land use, slightly lower for GWP and energy use, and much lower for eutrophication and acidification. Therefore, even when taking account of shipping transportation to the furthest market, NZ cheese appears to have either similar or less impact than its locally produced equivalent. However, it should be noted that this statement is based only on initial comparison between results from four separate studies at national level. A comparative study with harmonised methodology is needed to validate this analysis. Furthermore, more site-dependent impact assessment could affect the results of this study, particularly for acidification and eutrophication where actual impacts depend upon the sensitivity of local ecosystems to emissions.

4 CONCLUSION

The cradle to grave LCA of NZ cheese has been performed and enabled identification of the hot spots of this product across its life cycle stages. The farm stage was the main contributor to all impact categories, while sewage treatment was the second most important for eutrophication; the dairy manufacturing stage was the second most important for energy use. The transportation stages all together constituted a hot spot in terms of energy use for the life cycle of cheese at 38% of the overall energy use. For the other impacts, the transportation contribution was about 10% or less. However, across all transportation stages, shipping contributed a minor part of the overall energy use for transportation of cheese at 27%. A sensitivity analysis showed that NZ cheese exported to the UK had similar (GWP, energy use, land use) or less environmental impacts (eutrophication, acidification) than UK cheese, and less impacts than Swedish and Dutch cheese, even when accounting for the shipping stage. This first analysis needs to be validated with a detailed cradle to grave assessment for UK products and other EU products using harmonised methodology, accounting for variations between dairy farms within countries, and taking account of site-dependent impact assessment issues.

REFERENCES

1. Paxton, A., 1994. The Food Miles Report: The dangers of Long Distance Food Transport. The SAFE Alliance, London, UK.
2. Ministry of Agriculture and Forestry, 2005. Situation and outlook for New Zealand Agriculture and Forestry 2005. Ministry of Agriculture and Forestry, Wellington, New Zealand.
3. Statistics New Zealand, 2005. New Zealand External Trade Statistics. December 2005. Available at: www.stats.govt.nz/externaltrade
4. Basset-Mens, C., Ledgard, S., Boyes, M., Eco-efficiency of increasing scenarios of milk production in New Zealand. Submitted to Journal of Ecological Economics.
5. Lovell-Smith, J.E.R. Baldwin, A.J., 1988. "Energy Use Trends in the New Zealand Dairy Industry." New Zealand Journal of Dairy Science and Technology 23: 239-255.
6. LCAFood, 2003. LCA Food Database. Available at: http://www.lcafood.dk/
7. Sonesson, U., Janestad, H., Raaholt, B., 2003. Energy for preparation and storing of food. SIK rapport Nr 709 2003. SIK, Gothenburg, Sweden.
8. maritimeChain, 2006. Ports and Distances tool on www.maritimechain.com/port/port_distance.asp
9. Smith, A., Watkiss, P., Tweddle, G., McKinnon, A., Browne, M., Huntm A., Trelevenm C., Nash, C., Cross, S., 2005. The validity of Food miles as an Indicator of Sustainable development: Final report. Report for DEFRA. AEA Technology Environment, Didcot, UK.
10. Cederberg. C., Mattsson, B., 2000. Life cycle assessment of milk production – a comparison of conventional and organic farming. J. Cleaner Prod., 8:49-60.
11. Thomassen, M., van Calker, K.J., Smits, M.C.J., Iepema, G.J., de Boer, I.J.M., 2007. Life cycle assessment of milk production systems in the Netherlands. Agr. Syst., in press.
12. Williams, A.G., Audsley, E., Sandars, D.L., 2006. Determining the environmental burdens and resource use in the production of agricultural and horticultural commodities. Main report. Defra Research Project IS0205. Bedford: Cranfield University and Defra. Available at: www.silsoe.cranfield.ac.uk, and www.defra.gov.uk

All salmon are not created equal: the life cycle environmental impacts of salmon fisheries and culture in the NE Pacific

Presenter: Peter Tyedmers[1]
Co-authors: Nicole Arsenault[1], Nathan Ayer[1], Anna Flysjo[2], Sarah Kruse[3], Nathan Pelleter[1], Astrid Scholz[3], Ulf Sonesson[2]

Affiliation: 1. School for Resource and Environmental Studies
2. SIK, the Swedish Institute for Food and Biotechnology
3. Ecotrust

Abstract
Salmon are one of the most widely consumed seafood products globally. Although most current environmental concern regarding salmon production activities focus on largely proximate biological impacts – stock declines, by-catch, habitat damage, disease and potential genetic impacts, this focus overlooks the diverse environmental impacts that flow from the interlinked series of industrial activities that characterize most modern salmon fishing and farming systems. This presentation describes the results of an international life cycle assessment project to evaluate the impacts associated with major salmon fishing and farming activities of the NE Pacific. This includes those associated with Alaskan troll, purse seine and gillnet fisheries along with both conventional net-pen and experimental land-based culture systems based in British Columbia. Key findings include: impacts vary by an order of magnitude between fishing gears; although impacts associated with feed provision dominate within farming systems, the adoption of land-based culture technologies to address local ecological effects can markedly exacerbate global impacts; and differences in the location of primary production results in substantial impacts as a result of both the mix of primary energy availability and waste utilization. Opportunities for improving the environmental performance of both capture and culture systems will also be discussed.

Context
Salmon are one of the most widely consumed seafood products globally. Although historically, capture fisheries supplied all salmon consumed, over the last three decades, intensive salmon culture (salmon farming) has grown to the point that it now accounts for most of the world's production. Of the approximately 2.5 million live weight tonnes of salmon available globally in 2005, almost 1.6 million tonnes came from farms (FAO 2007). One region that hosts both substantial wild capture and farmed salmon industries is the NE Pacific. In 2005, Alaskan salmon fisheries landed about 400,000 tonnes while fisheries in British Columbia (BC) accounted for another 30,000 tonnes and salmon farming in BC produced 63,000 tonnes (FAO 2007).

Production Systems
Salmon production systems of the NE Pacific are highly diverse. Directed fisheries employing purse seine, gillnet or troll fishing gear account for the vast majority of commercial salmon landings. Of the five species of salmon targeted by commercial fisheries of the NE Pacific, pink and sockeye salmon (*Oncorhynchus gorbuscha* and *nerka* respectively) typically account for two thirds of total landings with chum salmon (*Oncorhynchus keta*) contributing another 10 to 20% and coho and chinook salmon (*Oncorhynchus kisutch* and *tshawytscha* respectively) making up the balance. Although there are a wide range of culture techniques used to artificially enhance the number of salmon available for harvest, most salmon caught are either wild spawned or are released from hatcheries when very small (as is the case with the major pink salmon hatchery enhancement that occurs in Alaska).

Salmon farming in the NE Pacific region is typical of the global industry. The entire life history of the animal occurs in culture. Eggs are hatched and juveniles are reared in freshwater before being transferred to marine-based net-pen structures typically located in sheltered coastal waters. Salmon are raised on a complete diet that is typically composed of approximately 60% animal-derived components (fish meal and oil and smaller fractions of feather, blood or meat meal from animal processing) and the balance comprised of various plant meals, oils or whole grains. Approximately three quarters of salmon farmed in the NE Pacific are Atlantic salmon (*Salmo salar*) with chinook accounting for most of the balance (FAO 2007).

In response to concerns regarding a range of environmental impacts associated with conventional net-pen based salmon farming, alternative culture technologies have been experimented with in BC including grow-out in land-based concrete tanks and marine-based pen structures in which nets are replaced with impervious bags.

Both farmed and wild caught salmon are processed into a variety of product forms. Those common to both include fresh, frozen and smoked fillets.

Our Analysis
Life cycle assessment (LCA) was used to evaluate and compare the environmental impacts associated with:

- typical Alaskan salmon fishing using purse seine, drift gillnet and troll fishing gears,
- conventional farmed salmon production in BC,
- use of alternative feed formulations and grow-out culture technologies, and
- processing of salmon into three common product forms (fresh, frozen and smoked fillets) in the vicinity of harvest and their delivery to the consumer in various locations in the United States.

Major LCA Results
Results of analyses are currently under review or are in preparation for submission. However, preliminary results include the following:

a) Alaskan salmon fisheries

Broadly consistent with prior LCAs of fisheries (Ziegler et al. 2003, Hospido and Tyedmers 2005), the vast majority (typically over 85%) of life cycle potential impacts result from direct fuel inputs regardless of fishing gear used (Figure 1). Not surprisingly therefore, major differences in the environmental performance between gear sub-fleets result from the substantial differences in direct fuel inputs. Consequently, troll fishing boats result in an order of magnitude greater emissions than do typical purse seiners per tonne of salmon landed (Figure 1).

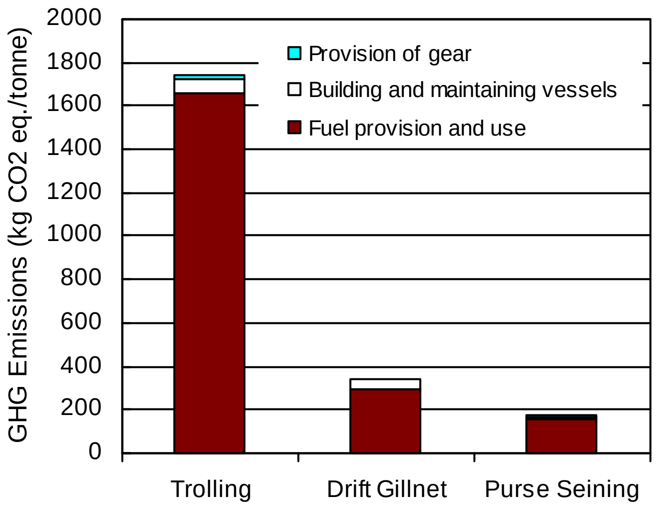


Figure 1 Gear specific life cycle greenhouse gas emissions up to the wharf per tonne of salmon landed in Alaska

b) BC salmon farming

Reflecting patterns described by earlier researchers (Papatryphon et al 2003 Aubin et al. 2006, Gronroos et al. 2006), the provision of feeds results in the vast majority of life cycle impacts associated with conventional salmon farming in BC up to the point of harvest. Although life cycle burdens association with the provision of individual feeding stuffs vary widely, in general, animal-derived inputs have a greater impact intensity (impact per unit mass of feeding stuff) than do their plant-derived equivalents. As a result, relatively substantial environmental performance improvements can be made to farmed salmon production through the partial substitution of plant- for animal-derived feed inputs. Interestingly, the substitution of organic for conventional crop inputs to feed did not result is as great an environmental performance improvement (Pelletier and Tyedmers in review).

Despite the dominant role that feed provision plays in the life cycle environmental impacts associated with conventional salmon farming, the use of some alternative grow-out technologies can result substantial increases in overall impacts. In particular, the increased use of electricity to pump and oxygenate water in land-based tank culture systems resulted in a roughly 40% increase in potential impacts up to the point of harvest (Figure 2). Importantly, the scale of these electricity-related impacts are much higher again if the primary energy inputs used to model the systems reflect the Canadian average energy mix as opposed to the that of British Columbia where 90% of electricity is hydroelectric (Figure 2) (Ayer and Tyedmers in review).

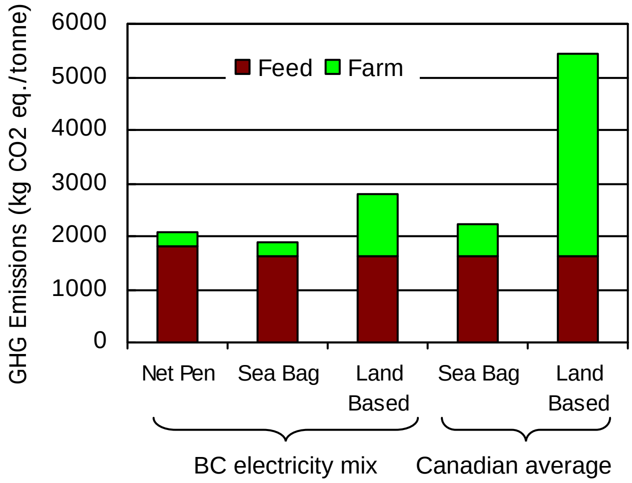
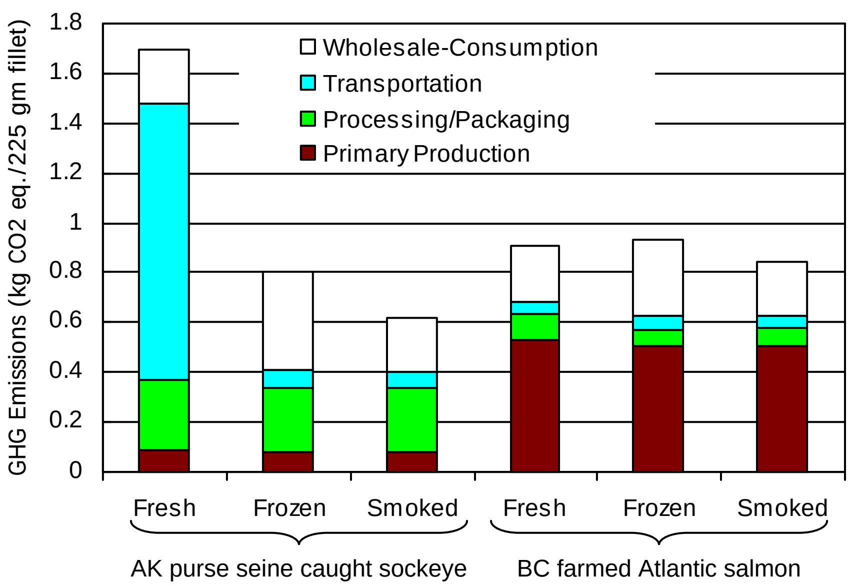


Figure 2 Greenhouse gas emissions associated with alternative salmon farming technologies used in BC and alternative primary energy mixes

c) Salmon processing

Despite the fact that freezing or smoking a salmon fillet requires more direct resource inputs than leaving it fresh, these differences result in relatively trivial differences in environmental impacts when compared with those associated with where the processing takes place. The direct implication of processing locale flows again from the primary energy mix associated with electricity generation. Wild caught salmon processed in Alaska depend on an electricity mix derived primarily from fossil fuels. This stands in stark contrast with the primarily hydroelectric system in British Columbia where farmed salmon are processed. The indirect impact of processing locale and form results from transport. In order to deliver fresh salmon fillets from Alaska to the lower 48 states requires the use of air transport that consequently results in a substantial increase in overall life cycle impacts (Figure 3).

Figure 3 Cradle to fork greenhouse gas emissions associated with provision of 225 gm fresh, frozen and smoked fillets to consumer in San Francisco



Conclusions

Both how and where salmon are "produced" can have substantial impacts on the overall life cycle impacts associated with a typical salmon fillet delivered to the consumer. Fuel and feed inputs are key drivers in typical salmon fishing and farming systems respectively. Post-harvest, life cycle impacts are heavily influenced by the primary energy mix used to generate electricity and the mode of transport used, both factors that are largely influenced by the location of production.

References

Aubin, J., E. Papatryphon, H. Van der Werf, J. Petit and Y. Morvan. 2006. Characterization of the environmental impact of a turbot (*Scophthalmus maximus*) re-circulating production system using Life Cycle Assessment. *Aquaculture* 261(4):1259-1268.

Ayer, N., and P. Tyedmers (in review) Life cycle environmental impacts of alternative salmonid farming technologies in Canada. Submitted to *Journal of Cleaner Production*.

Gronroos, J., J. Seppala, F. Silvenius and T. Makinen. 2006. Life cycle assessment of Finnish cultivated rainbow trout. *Boreal Environmental Research* 11(5):401-414.

Hospido, A., and P. Tyedmers, (2005) Life cycle environmental impacts of Spanish tuna fisheries. *Fisheries Research* 76(2):174-186.

Papatryphon, E., J. Petit, H. Van der Werf and S. Kaushik. 2003. Life Cycle Assessment of trout farming in France: a farm level approach. Life Cycle Assessment in the agri-food sector. *Proceedings from the 4th International Conference Dias Report* 61:71-77.

Pelletier, N and P. Tyedmers (in review) Feeding farmed salmon: Is organic better? Submitted to *Aquaculture*.

Ziegler, F., P. Nilsson, B. Mattsson, and Y. Walther (2003) Life Cycle Assessment of frozen cod fillets including fishery-specific environmental impacts. *Int. J. LCA* 8(1):39-47.

Can LCA studies improve the environmental impact of the fisheries?

Harald Ellingsen
Dept. of Marine Technology, Faculty of Engineering Science and Technology, Norwegian University of Science and Technology (NTNU), NO-7491 Trondheim, Norway

Abstract
Several LCA studies performed within the marine sector have analyzed the fish from the catch phase to the end consumer in order to evaluate environmental impacts. In general it is concluded that the catch phase is the "hot spot", due to the energy consumption, sea floor disturbance, ghost fishing, and bycatch. However, it may be questioned to what degree these conclusions affect the development of the fisheries industry. The fishing gears are getting larger, heavier, and require more installed power to be operated properly, but otherwise not much have changed the last decades. In recent years, environmental labels have been introduced, but mostly these are motivated by the fisheries management objective of preventing overexploitation, and not by the way the fish is caught. In this paper various ways to communicate the results of some Norwegian LCA studies are discussed in order to improve the environmental impact of the fisheries industry.

1. INTRODUCTION

The fisheries is an important industry in Norway. The Norwegian coastal areas are among the worlds most productive, and Norway is one of the world's largest exporters of seafood. In 2005 about 90% of the Norwegian seafood was exported, valued NOK 31, 7 billion [1]. The production capacity of the sea relies on balanced harvesting, pollution control etc, and the fisheries industry needs a market willing to pay high prices for the products. Thus, the demand for documentation that proves that the sea food is harvested, produced, and processed according to sound environmental principles is increasing [2]. In this context, important issues concern defining sustainable sea food productions, finding ways to "prove" that the sea food production is environmental friendly, and evaluating if LCA is an applicable method for supplying relevant information regarding these issues.

Several environmental impacts from the fisheries are identified, such as overexploitation, overcapacity, high energy consumption, sea floor disturbances, emissions of heavy metals, ghost fishing, bycatch and bymortality, as well socio-economic issues regarding the high numbers of occupational accidents and fatalities. Some fisheries impose more severe environmental impacts than others, but it can hardly be argued that any fishing activity can take place without any environmental impact at all. Overcapacity is a fundamental problem in many global fishing communities straining the management organizations around the world [3]. Environmental challenges can also be identified for other parts of the sea food industry, such as aquaculture [4].

This paper describes LCA and other environmental analyses that have been applied in the Norwegian fisheries, and discusses the efficiency so far of these evaluations to improve the environmental performance of the industry.

2. LCA APPLIED IN THE FISHERIES

In the fisheries, the LCA methodology is used to deal with some of the environmental impacts described in the introduction. A short description of status of the availability of the methods can be found in table 1.

Table 1: Status of LCA used in the fishing industry. Adapted from [5].

Environmental impacts	LCA methods – state of the art
Overexploitation	Not included (should be avoided in any case)
Energy use	Included in LCA analyses
Anti fouling	Included in LCA analyses
Eutrophication	Included in LCA analyses
Material use	Included in LCA analyses
Sea bed effects	Methods developed for land use
Safety	Methods developed for industrial processes
Bycatch	To be developed
Discards	To be developed
Ghost fishing	To be developed

In discussions with fishermen, fishermen's organizations, fishing gear producers, environmental groups, and representatives from fisheries management, researchers are often confronted with questions about the fisheries industry's environmental performance and possible means of efforts to reduce impact. LCA information is not always easy to understand to "outsiders", and the results are not always very conclusive. If the fisheries industry shall respond to the results from LCA analyses, the information should be possible to transfer into concrete terms, for example leading to technical solutions that reduce the fuel consumption of the fishing vessel engine and propulsion system.

From 2002 to 2006, SINTEF Fisheries and Aquaculture performed a strategic research program called "Sustainable fishing vessels and fleet structure" in which methods and strategies were developed to improve the environmental performance of the Norwegian fishing fleet. Use of the LCA methodology was one important part of this program. In the programme, the three main strategies carried out were:

1. To document the environmental impacts from the fisheries at an "overall" level in order to assess whether the fisheries industry is sustainable or not.
2. To measure the "sustainability level" of the fishing fleet at various time intervals in order to identify trends and possible means to reduce environmental impacts.
3. To improve the environmental performance of the individual components like the fishing gear, the fishing vessel and so on.

Strategy 1 was an attempt to supply information about environmental performance to relevant stakeholders. In this case an acceptable level of sustainability had to be defined, before identification of the current situation in the fisheries industry. A comparison between wild caught cod, farmed salmon and chicken was performed, concluding that some impact categories like energy consumption are possible to assess in a traditional LCA, but LCA can hardly be used to find an overall estimate for the collected level of sustainability in the fishing fleet [4].

In order to evaluate the problem of overcapacity, it is possible to calculate the technical catch capacity of a fleet or a fleet segment, and compare this with the available marine resources. These results are found without using LCA. Standal [6] has shown that the total catch capacity has increased even though the number of Norwegian fishing vessels has been reduced during the last decades. The catch capacity has especially increased within the largest vessels of the coastal and the ocean going fleet, mainly due to implementation of new technology. A sustainable acceptance level for the overall energy use or more precise emissions to air can also be defined based on political targets defined by Norwegian commitments to the Kyoto and Gothenburg protocols [7], [8].

In strategy 2, the measurements of the "overall" level of sustainability in the fishing fleet, should be carried out by use of performance indicators aiming at exploring trends. By studying the environmental performance of various fleet segments, Utne [9] has found that the smallest cod-fishing vessels are scoring better on most of the performance indicators defined, such as fuel consumption and earning capacity. The largest vessels however have a much lower accident risk level, and if stakeholders assess safety as the most important indicator, the largest vessels are more sustainable than the smallest vessels. She has also shown that performing such analyses at various time intervals, trend lines can be found with respect to sustainability.

As is the case in strategy 2, strategy 3 presupposes development of environmental indicators that are mutually accepted by relevant stakeholders. Such indicators must be related to measurements of the gear and vessel impact versus the food chain of the fish, and not only to the production of the technical component itself. That is, it is of no use to optimize the material use, the production process etc. of a fishing gear if it does not catch fish in an efficient way. Improvements can be identified with respect to sea bed contact, selectivity abilities, hydrodynamic resistance, risk level and working environment, use of energy carrier etc. In wide terms, this is the strategy level where it should be easy to communicate with fishermen, fishing gear producers, and fishing vessel designers to promote environmental improvements at the component level.

A good example is use of natural gas instead of diesel for propulsion. Use of natural gas improves the environmental performance of a fishing vessel by reducing the NOx (85%) and the CO2 (20%) emissions [10], and it is possible to link improvements at the component or vessel level to the fleet level, and further compare the results with political targets with respect to emission reductions. In other cases this link is however not so easy to find. There are good environmental reasons for replacing toxic anti fouling with less toxic alternatives, but this may lead to increased fouling and increased hull resistance and thus increased emissions to air.

3. CONCLUSION

Strategy 1 may produce information of interest to fisheries management, environmental groups, and fishermen's organizations. However, it will always be a political based evaluation whether the industry is environmental "acceptable" or not. The potential of strategy 2 is to give both the authorities and non-governmental organizations' assistance in assessing the environmental performance of the industry, by contributing more information about responds to changes in the management regimes over time. Nevertheless, strategy 3 should be the main level of communication with the industry, in order to make improvements on the component level which again will improve the environmental performance for the whole fleet or fleet groups.

LCA analyses have proved to be useful on this level even if there are still much to be done as the need for development of improved indicators adapted to the fishing industry. The need for better methods for the assessment of the environmental effects of bottom effects is one such example.

4. REFERENCES

1. The Norwegian Ministry of Fisheries and Coastal Affairs, *Facts about Fisheries and Aquaculture*. 2006: www.odin.dep.no (Accessed: 2006:09:06).
2. The Norwegian Ministry of the Environment, *Report no. 21 to the Storting (2004-2005): The government's environmental policy and the state of the environment in Norway*.
3. Food and Agricultural Organization of the United Nations, *The State of World Fisheries and Aquaculture 2004*. 2004: www.fao.org (Accessed: 2007:01:26).
4. Ellingsen H., A.A., *Environmental Impacts of Wild Caught Cod and Farmed Salmon – A Comparison with Chicken.* International Journal of Life Cycle Assessment, 2006. **11**(1): p. 60-65.
5. Thrane, M., *Environmental Impacts from Danish Fish Product. Hot spots and environmental policies*, in *Department of Development and Planning, Aalborg University, Denmark*. 2004.
6. Standal, D., *Nuts and bolts in fisheries management- a technological approach to sustainable fisheries?* Marine policy, 2005. **29**: p. 255-263.
7. The Norwegian Ministry of the Environment, *Report no. 29 to the Storting (1997-1998): Norway's follow up of the Kyoto Protocol*.
8. Norwegian Pollution Authority and the Ministry of the Environment, *A rift of sun in acid clouds. International agreements lead to cleaner Norwegian nature*. 2000.
9. Utne, I.B., *System evaluation of sustainability in the Norwegian cod-fisheries.* Marine Policy, in press, 2007.
10. Ellingsen, H., Lønseth, M., *Energy-reducing efforts in the Norwegian fisheries*. 2005: SINTEF Fisheries and Aquaculture.

Life-cycle assessment of Bio-diesel from *Jatropha curcas* L. energy balance, impact on global warming, land use impact

W. Achten[a], B. Muys[a, 1], E. Mathijs[b], V.P. Singh[c] and L. Verchot[d]

[a] *Division Forest, Nature and Landscape, K.U.Leuven, Celestijnenlaan 200 E, Box 2411, BE-3001 Leuven, Belgium.*
[b] *Division Agricultural and Food Economics, K.U.Leuven, de Croylaan 42, Box 2424 BE- 3001 Leuven, Belgium*
[c] *World Agroforestry Centre Regional office for South Asia, CG Block, 1st Floor, National Agricultural Science Centre, Dev Prakash Shastri Marg, Pusa, New Delhi - 110 012, India*
[d] *World Agroforestry Centre, Head Quarters, United Nations Avenue P.O. Box 30677-00100 Nairobi, Kenya*

Abstract
The pantropic oil-bearing woody plant *Jatropha curcas* L. (JCL) receives a lot of attention from Clean Development Mechanism project developers all over the tropical world. The crop as living fence is good for food production protection, erosion control and ecological restoration in degraded semi-arid regions. JCL is suitable for several agroforestry and intercropping cultivation systems as well. Nowadays JCL is widely planted as monoculture. Besides the cultivation, the production process of bio-diesel consists of extracting the oil from the seeds and conversion of the crude oil to bio-diesel. The production process results in a whole range of interesting by-products as well. At the moment no complete life cycle assessment of the bio-diesel production from JCL is available. The authors started research which will focus on such LCA, ivestigating the 3 different JCL cultivation systems using 2 reference systems and strengthened by a socio-economic impact study.

1. INTRODUCTION

Jatropha curcas L. (*Euphorbiaceae*) is a small deciduous tree (up to 5 m) which originates from Mexico and Central America, although nowadays is growing pantropic [1]. The multi purpose crop is traditionally used for medicinal purposes, but is also useful as living fence and for the prevention and control of soil erosion [2,3]. As a pioneer species, well adapted to semi-arid climates, JCL is promising to simultaneously combat desertification, produce bio-diesel and enhance socio-economic development in degraded rural areas in the South [4].

In normal conditions the plant will fruit once a year, yielding 2-5 tons of dry seed/ha/year, after 5 years, depending on the genetic variety, agro-climatical conditions and the management input [1,4,5]. The seeds contain 30-35 % oil by weight, which can easily be converted into bio-diesel meeting the standards of US, Germany and European Standard Organisation [6]. The seeds and oil are not edible due to presence of toxins as phorbol esters, trypsin inhibitors, lectins, phytates [4].

These properties persuade many Clean Development Mechanism project developers, investors and policy and opinion makers to choose JCL to tackle the energy dependency and greenhouse gas problems of fossil fuels, although the environmental impacts have not been investigated yet. There is a clear need for a LCA study of bio-diesel from *Jatropha*.

[1] Corresponding author. Bart Muys: Tel.: +32 (0) 16 329726; Fax: +32 (0) 16 329760;
E-mail address: bart.muys@biw.kuleuven.be

2. BIO-DIESEL PRODUCTION FROM JATROPHA

2.1. Jatropha cultivation

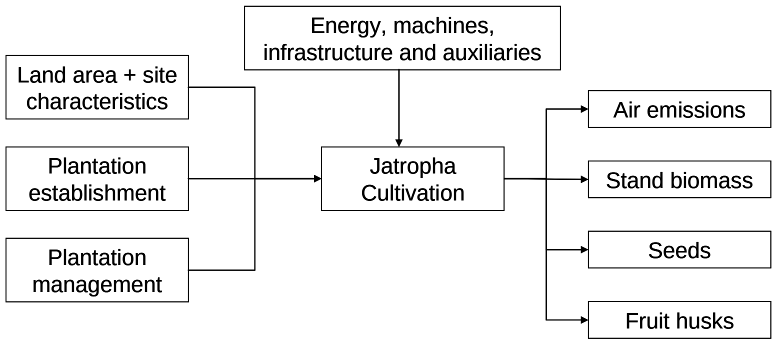


Figure 1. Flow chart of the *Jatropha* cultivation unit process

JCL's high adaptability [1] allows it to grow in wide ranges of conditions. As a succulent that sheds its leaves during the dry season, JCL is best adapted to arid and semi-arid conditions. [7]. All soil types can support JCL except for *Vertisols* and soil with pH above 9 [5,8].
For establishment of long living plantations for oil production, seed propagation is preferred above vegetative propagation (cuttings) [1]. Seedlings are prepared in nursery condition and planted in planting pits which are best refilled with a mixture of soil, compost, sand, organic matter and/or artificial fertilizer [9]. Further management includes pruning and canopy management, fertilizing and irrigating according to the situation [9-11]. Due to uneven ripening, the fruits are harvested manually [1]. Separation of the seeds and husks can be done manually or mechanically [9]. These fruit husks can be gasified [12], fermented for biogas production [13] or combusted directly. The air or oven dried seeds go to the following production step.

2.2. Oil extraction

For extraction of the JCL oil two main methods have been identified: (i) mechanical extraction and (ii) chemical extraction [14,15]. Most common is the use of an engine driven screw press, achieving a yield of 70-80% of the available oil [5,16,17]. Manual presses achieve 60-65% [5,15,16].

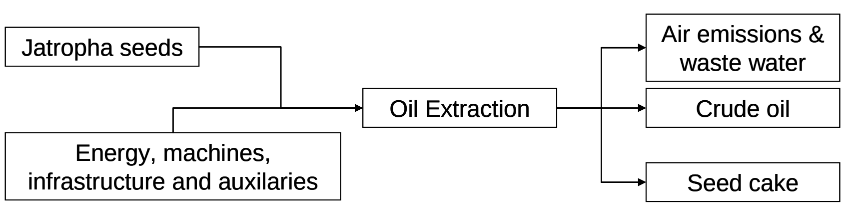


Figure 2. Flow chart of the oil extraction unit process

Solvent extraction (using *n*-hexane, acetone, agueous enzymatic oil extraction, …) generally achieve higher yields (up to 98% of available oil) [2], but is only economical viable at a large scale production [18]. Furthermore, the conventional *n*-hexane solvent extraction is believes to have a high specific energy consumption and high greenhouse gas emission [18]. After extraction the oil is filtered through a filter press and is then ready for the conversion to bio-diesel.
The seed cake left over after extraction is valuable as bio-fertilizer, as it contains more nutrients than both chicken and cattle manure [4]. Before using the cake as fertilizer, the cake can serve as feed for biogas production as well [19,20].

2.3. Transesterification

Transesterification is the displacement of alcohol from an ester by another alcohol [21]. In case of JCL oil, (m)ethanol displaces glycerol from triglycerides, leaving (m)ethylesters (i.e. bio-diesel).
In order to shift the reaction to the right an alcohol excess (molar ratio alcohol:oil = 6:1) and a catalyst (NaOH, KOH at 20% by weight on oil basis) are necessary [22]. An optimal ester yield of 98% is achieved after 90 min of reaction at 60°C [22]. Crude glycerol is separated an can be used as a raw material for soap production or other cosmetica.

3. RESEARCH OBJECTIVES

An independent, generic, comparative LCA will be made, comparing 3 different JCL production systems, using 2 reference systems: (i) an other tropical bio-diesel system and (ii) the conventional fossil diesel system (figure 4).). The functional unit where all outputs and inputs will be related to is '100km driven with a 4×4 pick up'. Energy balance, global warming potential and land use impact are seen as the most relevant impact categories.

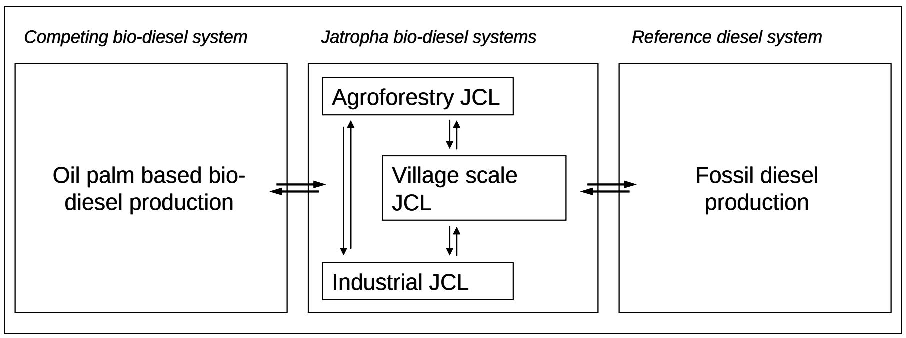


Figure 4. LCA comparison objectives

4. PRELIMINARY RESULTS AND EXPECTATIONS

4.1. Energy balance

The energy balance is reported to be positive [23,24]. In both reports transesterification is shown to be an energy intensive production step. The big difference between the two studies lies in the share of the jatropha cultivation in the overall energy input (50 in [23] and17% in [24]). This shows the importance of the applied cultivation intensity. Fertilizer and irrigation are energy intensive agricultural practices.

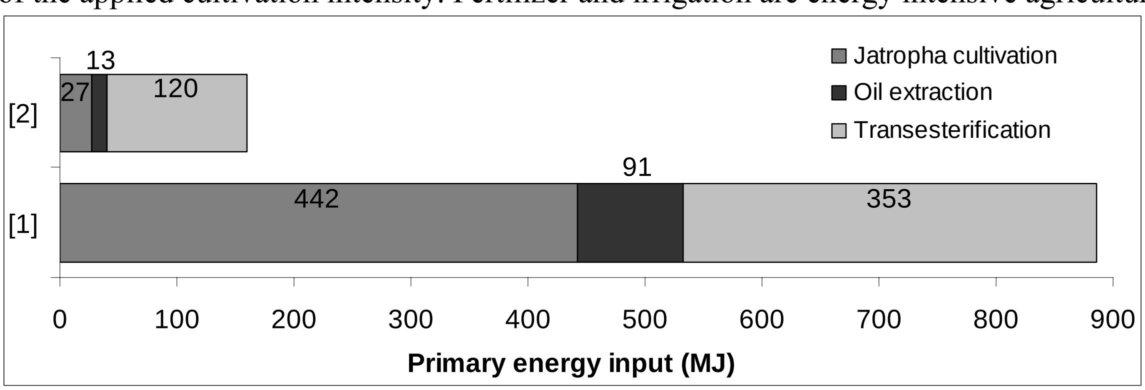


Figure 5. Primary energy input for the production of 1000 MJ *Jatropha* bio-diesel. Based on: [23] with high cultivation input and [24] with low cultivation input system.

Based on these reports it is believed that the energy balance of bio-diesel from *Jatropha* is positive but is thought to become less positive after:

- After transesterification

- Without energetic use of the by-products
- With increasing intensification and mechanization of the production cycle
- If the product is shipped to remote markets (such as Europe)

4.2. global warming potential

Tobin & Fulford [24] and Prueksakorn & Gheewala [23] also showed positive results on the life cycle greenhouse gas emission of the production of *Jatropha* bio-diesel in comparison to fossil diesel (figure 6). Again there is a clear difference in the obtained results of a high cultivation input (described in [23]) and low cultivation input system (figure 6).

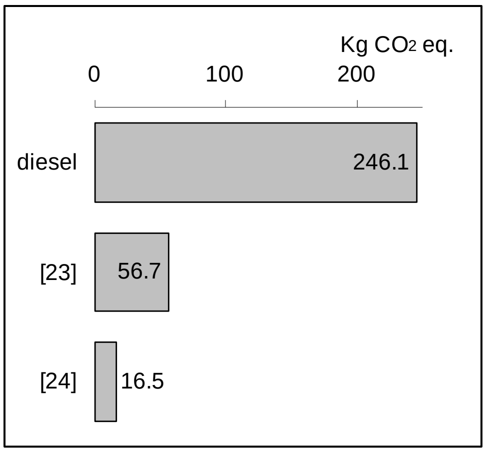


Figure 6. Life cycle greenhouse gas emissions for the production of 1000 MJ *Jatropha* bio-diesel in comparison with fossil diesel. [23] with high cultivation input and [24]with low cultivation input system.

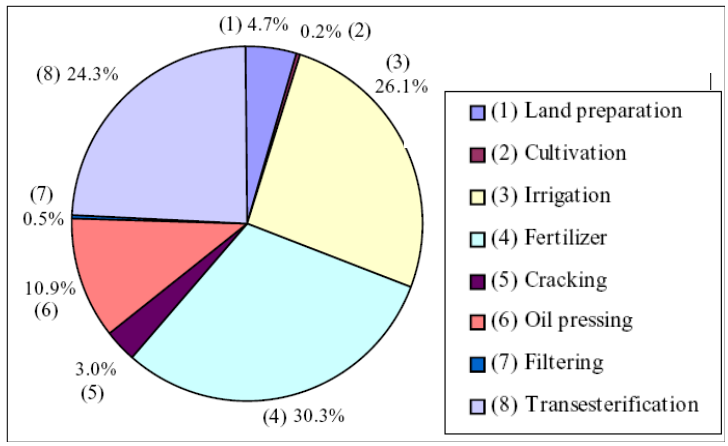


Figure 7. Representative greenhouse gas emissions of the production steps in the *Jatropha* bio-diesel production process. *Source*: [23]

Figure 6 and 7 indicate that the impact on the global warming potential is positive. But the impact becomes less positive with the intensification of the *Jatropha* cultivation (mainly fertilizer and irrigation application are greenhouse gas intensive inputs). Furthermore it is clear that also the transesterification is a big greenhouse gas contributor.

4.3. Land use impact

The land use impact assessment will be made, based on the method described by Peters et al. [25]. 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: soil, water, vegetation structure and biodiversity. Thematic scores are multiplied by the area x time needed for the production of the functional unit [25].

Expected results for *Jatropha* bio-diesel:

- Impact on soil: (i) Mostly positive impact through erosion control and carbon sequestration, but (ii) negative impact if intensively grown with high input of fertilizers and machinery
- Impact on water: based on Heuvelmans et al. [26]: positive on-site effects, but negative off-site (more research necessary)

- Impact on vegetation structure: (i) Positive if wasteland is reclaimed, (ii) negative in case of re-allocating (semi-)natural vegetation to *Jatropha*.
- Impact on biodiversity: (i) negative in monoculture. Improvement possible by intercropping, agroforestry and set aside part of the land. (ii) Positive in case of low use of biocides. (iii) Further there are some unchecked reports on invasiveness.

4.4. Other impact categories

Since JCL seeds and oil contain several toxins, such as phorbol esters, curcin, trypsin inhibitors, lectins, phytates, to such levels that the seeds and the oil are toxic to human Attention should be paid to human health and work environment. These impact categories are also at stake in case of solvent extraction.

5. CONCLUSIONS

Jatropha curcas L. is a promising energy crop for the semi-arid regions. Preliminary results show a positive energy balance and impact on global warming potential. More research is necessary to get a good insight in the environmental sustainability of this production system. The land use impact is an absolute must to address those sustainability issues.

6. ACKNOWLEGMENTS

This research is funded by the Flemish Interuniversity Council – University Development Co-operation (VLIR-UOS) and is a collaboration between K.U.Leuven and ICRAF Nairobi, Kenya and New Delhi, India

7. REFERENCES

1. J. Heller
   Physic nut. *Jatropha curcas* L. Promoting the conservation and use of underutilized and neglected crops. 1.
   Ph.D Dissertation.
   Rome: Institute of Plant Genetic and Crop Plant Research, Gatersleben, Germany & International Plant Genetic Resource Institute, Rome, Italy, 1996.
2. G.M. Gubitz, Mittelbach M. and Trabi M.
   Exploitation of the tropical oil seed plant *Jatropha curcas* L.
   Bioresource Technology, 67(1): 73-82,1999.
3. R.K. Henning
   The Jatropha System.
   http://www.jatropha.de/
4. G. Francis, Edinger R. and Becker K.
   A concept for simultaneous wasteland reclamation, fuel production, and socio-economic development in degraded areas in India: need, potential and perspectives of *Jatropha* plantations.
   Natural Resources Forum, 29(1): 12-24,2005.
5. D.N. Tewari

Jatropha & Biodiesel.
228 p., Ocean Books Ltd., New Delhi, 2007.

6. M.M. Azam, Waris A. and Nahar N.M.
Prospects and potential of fatty acid methyl esters of some non-traditional seed oils for use as biodiesel in India.
Biomass & Bioenergy, 29(4): 293-302,2005.
7. N. Foidl, Foidl G., Sanchez M., Mittelbach M. and Hackel S.
Jatropha Curcas L. as a Source for the production of biofuel in Nicaragua.
Bioresource Technology, 58(1): 77-82,1996.
8. S. Biswas, Kaushik N. and Srikanth G.
Biodiesel: technology and business opportunities - an insight
Proceedings of the biodiesel conference toward energy independance - Focus of *Jatropha*: 303-330, Edited by B. Singh, Swaminathan R. and Ponraj V., Hyderabad, India.
9. V.K. Gour
Production practices including post-harvest management of *Jatropha curcas*
Proceedings of the biodiesel conference toward energy independance - Focus of *Jatropha*: 223-251, Edited by B. Singh, Swaminathan R. and Ponraj V., Hyderabad, India.
10. L. Singh, Baragali S.S. and Swamy S.L.
Production practices and post-harvest management
Proceedings of the biodiesel conference toward energy independance - Focus of *Jatropha*: 252-267, Edited by B. Singh, Swaminathan R. and Ponraj V., Hyderabad, Indi.
11. N. Kaushik and Kumar S.
Jatropha curcas L. silviculture and uses.
Agrobios, Jodhpur, 2006.
12. D.K. Vyas and Singh R.N.
Feasibility study of *Jatropha* seed husk as an open core gasifier feedstock.
Renewable Energy, 32(3): 512-517,2007.
13. O. Lopez, Foidl G. and Foidl N.
Production of biogas from *J. curcas* fruitshells.
Biofuels and industrial products from *Jatropha curcas* - Proceedings from the Symposium "Jatropha 97": 118-122, Edited by G.M. Gübitz, Mittelbach M.and Trabi M., Managua, Nicaragua,
14. A.O. Aderibigbe, Johnson C.O.L.E., Makkar H.P.S., Becker K. and Foidl N.
Chemical composition and effect of heat on organic matter- and nitrogen-degradability and some antinutritional components of *Jatropha* meal.
Animal Feed Science and Technology, 67(2-3): 223-243,1997.
15. F.K. Forson, Oduro E.K.and Hammond-Donkoh E.
Performance of *Jatropha* oil blends in a diesel engine.
Renewable Energy, 29(7): 1135-1145,2004.
16. R.K. Henning
The Jatropha Booklet - A guide to the *Jatropha* system and its dissemination in Zambia.
bagani GbR, Weissensberg, 2000.

17. E.L.M. Rabé, Somers L.M.T. and Goey L.P.H.
Jatropha oil in copression ignition engines - Effects on the engine, environment and Tanzania as supplying country.
Msc Dissertation.
Eindhoven, The Netherlands: Eindhoven University of Technology, 2005.
18. T. Adriaans
Suitability of solvent extraction for *Jatropha curcas.*
Eindhoven: FACT Foundation, 2006.
19. R. Staubmann, Foidl G., Foidl N., Gübitz G.M., Lafferty R.M. and Valencia Arbizu V.M.
Production of biogas from *J. curcas* seeds press cake.
Biofuels and industrial products from *Jatropha curcas* - Proceedings from the Symposium "Jatropha 97": 123-131, Edited by G.M. Gübitz, Mittelbach M. and Trabi M., Managua, Nicaragua,
20. P. Radhakrishna
Contribution of de-oiled cakes in carbon sequestration and as a source of energy, in indian agriculture - need for a policy initiative
Proceedings of the 4th international biofuels conference: 65-70, Edited by B.A. Kumar and Paul S., New Delhi, India,
21. L.C. Meher, Vidya Sagar D. and Naik S.N.
Technical aspects of biodiesel production by transesterification - a review.
Renewable and Sustainable Energy Reviews, 10(3): 248-268,2006.
22. P. Chitra, Venkatachalam P. and Sampathrajan A.
Optmisation of experimental conditions for biodiesel production from alkali-catalysed transesterification of *Jatropha curcas* oil.
Energy for Sustainable Development, 9(3): 13-18,2005.
23. K. Prueksakorn and Gheewala S.H.
Energy and greenhouse gas implications of biodiesel production from *Jatropha curcas L.*
Proceedings of The 2nd Joint International Conference on "Sustainable Energy and Environments (SEE 2006)": Anonymous., Bangkok, Thailand,
24. J. Tobin and Fulford D.J.
Life Cycle Assessment of the production of biodiesel from Jatropha.
Msc Dissertation.
The University of Reading, 2005.
25. J. Peters, Garcia-Quijano J., Content T., Van Wyk G., Holden N.M., Ward S.M. and Muys B.
A new land use impact assessment method for LCA: theoretical fundaments and field validation.
Life Cycle Assessment in the Agri-food sector. Proceedings from the 4th International Conference: Anonymous., Bygholm, Denmark,
26. G. Heuvelmans, Muys B. and Feyen J.
Extending the life cycle methodology to cover impacts of land use systems on the water balance.
International Journal of Life Cycle Assessment, 10(113-119,2005.

Coexistence scenarios between GM and GM-free crops

E. Balducci[a], M. Mazzoncini[a] and S. Gorelli[b]

[a]Pisa University, Department of Agronomy and Agro-ecosystem Management, Via San Michele degli Scalzi, 2 - 56124 Pisa, Italy.
[b]Pisa University, Department of Civil Engineering, Via Diotisalvi, 2- 56126 Pisa, Italy.

Abstract
The introduction of GM-crops has strong consequences at environmental and socio-economic level. Through the elaboration of Coexistence Plans, parameters will be identified to allow the coexistence of every kind of cultivation systems in the same environment (conventional, organic, GMO). This study investigates, by Life Cycle Assessment (LCA), environmental pressures coming from GM-corn cultivation in comparison with traditional corn, in connection with its socio-economic repercussions. LCA is a useful tool to represent productive process activity, but it shows some limits. Nevertheless, LCA results are useful to support scenarios analysis reaching the conclusion that GMO introduction conduces in the direction of major public and private costs.

1. INTRODUCTION

Coexistence concept is expressed in 2003/556/CE Commission Recommendation with these words: "*No form of agriculture be it conventional, organic, or agriculture using GMOs, should be excluded in the European Union*". The authorised GM crops for cultivation in UE are corn and rap; in the territory under study (Tuscany – Italy), corn is cultivated on considerable surfaces (23.856 ha in 2006), while rape is not common; for this reason we chose corn as reference crop of our analysis. The aim of this study was the assessment of environmental and economic aspects to define coexistence scenarios of GM and non-GM crops in Tuscany. Their environmental and energetic impacts have been evaluated by a screening *LCA*, while the economic impacts have been investigated by the *economic analysis*. The assessment of both environmental and economic aspects allows analysing the different scenarios that probably occur in the reality through the development of *scenario analysis* that may be an useful tool to define the *Regional Coexistence Plans* in conformity with Directive 2001/18/CE. In this work, we have evaluated the corn supply chain where Agricultural phase is considered the segment more interesting from both environmental and economic point of view. From the LCA analysis point of view, agricultural phase is the only one that presents significant differences in the productive process between GM-corn and conventional corn. From the side of the economic analysis, GM-corn and conventional corn cultivation are characterized by more differences in terms of sustainable costs and perceptible income at farm level.

2. MATERIALS AND METHODS

This study is based on *Scenario analysis* performed with the contribution of LCA and economic analysis. The utilisation of scenario analysis for the investigation of coexistence effects is widely documented (Bock *et al.*, 2002; Messéan *et al.*, 2003; Messéan *et al.*, 2006; INRA, 2004-2007) and the use of this analytical tool to support the scenarios implementation is also documented (Gaugitsch H., 2002; Theodosiou G. *et al.,* 2005). In particular, LCA analysis has been used to evaluate productive processes and to assess environmental and energetic impacts of the agricultural systems; LCA was performed with GaBi software (http://www.gabi-software.com/). The use of LCA to evaluate the effects of environmental impacts of growing GM crops is just known (Bennett R. *et al.,* 2004). Economic analysis has been utilized to evaluate the economic performance of the whole corn supply chain. The functional unit of the study was one hectare of production, and the system boundaries were the agricultural phase.

3. RESULTS AND DISCUSSION

1. LCA analysis – Within the whole corn supply chain, Agricultural phase conventionally consists of the following steps: ploughing, seed-bed preparation, fertilizing, sowing and geodisinfestation, herbicide, insecticide and fungicide treatments, mechanical cultivation, irrigation, harvest and stubble cutting up. Some phases of these can be modified introducing Bt GM corn (where Bt means *Bacillus thuringensis*) resistant to *Ostrinia nubilalis*, or RR GM corn (where RR means Roundup Ready) resistant to *Glyphosate* herbicide (Round-up). In our analysis we have taken in consideration this last case, because in central Italy *Ostrinia nubilalis* control is not so common among farmers. Introducing RR corn, weed control strategy changes deeply at farm scale: in comparison with the conventional strategy based upon pre- and post-emergence herbicide treatments (specifically *Terbutilazine* as pre-emergence herbicide and *2,4-D + MCPA* as post-emergence one), sowing GM corn, it is possible to cut-off any pre-emergence treatment and to control weed flora simply in post-emergence using *Glyphosate.*
In performing the screening LCA, the following assumptions have been considered: (i) *Quantitative data:* lack of objective data (i.e. yields, chemical inputs, etc.) due to the subjectivity of agricultural applications and the insufficiency of data relating to reference historic series; (ii) *Herbicide productive data:* absence of data relating to the considered chemical substances (including in GaBi database); consequent necessity to recourse to a strictly energetic evaluation of the herbicide production impact utilizing the unique conversion index available that estimates the energy necessary for their production(Pimentel, 1980); (iii) *Herbicide type:* assumptions due to the lack of herbicide products more recent in literature (i.e. utilization of *Atrazine*, nowadays forbidden, instead of *Terbutilazine*); the absence of herbicide mixture and consequently utilizing of less selective products; (iv) *Chemicals dispersion:* we assumed that all the herbicide go in the ground; (v) *Herbicide quantity for hectare:* in our study it corresponds to the principle active quantity, data about the additives production are not included; (vi) *Impact assessment:* at the moment, LCA hasn't standardized impact categories related to ecological and human health effects connected with GM plants (toxin toxicity, genetic pollution, biodiversity reduction, herbicide resistance increase). As regard the impact assessment, we decided to analyse only the "Global Warming Potential" because we evaluated the herbicide production process strictly from the energy consumption point of view.

Table 1: LCA analysis - CML2001, Global Warming Potential (GWP 100 years) [kg CO_2-Equiv.]

GM-free		GM	
Total	79,66	Total	39,80
2,4-D-MCPA	0,03	Glyphosate	0,01
Terbutilazine	0,06		
Diesel	78,81	Diesel	39,41
Lubrificant oil	0,76	Lubrificant oil	0,38

Source: our elaboration with GaBi software

The estimation concerning the other impact categories (Abiotic Depletion, Acidification Potential, Eutrophication Potential, Ozone Layer Depletion Potential, Photochemical Ozone Creation Potential, Radioactive Radiation) didn't give significant results. The element that gives the highest contribution to Global Warming is diesel consumption, higher in the case of GM-free cultivation, due to the two herbicide treatments needed in comparison with the GM corn. Table 1 shows the contribute to Global Warming of the different elements analysed: herbicides, diesel and lubrificant oil.

2. Economic analysis - In the framework of corn supply chain, agricultural phase represents the step with more differences in terms of costs and incomes between GMO and GMO-free farms. The detailed analysis of the different cost categories involved in the coexistence maintenance at agricultural level (Figure 1) shows that the higher GMO farms direct costs are due to assurance costs (e.g.. for the contamination of conventional production) and to the costs for the introduction of buffer zone of traditional corn around GMO area to contain the contamination phenomena. Other kind of costs are in charge of GMO-free farms, because are essentially measures of their directly application to remove the risk of contamination and to have the certainty of GMO-free productions. The necessity to support these costs determines notable economic, management, ethic unease to GMO-free farmers. The coexistence principle means that the farmer must be free to choose if cultivate GM crops, but he must be free to decide to cultivate non GM crops too. For this reason, the only way to give equal repartition of farm burdens is the transfer of this extra costs from GMO farms to GMO-free farms, how is shown in the figure below.

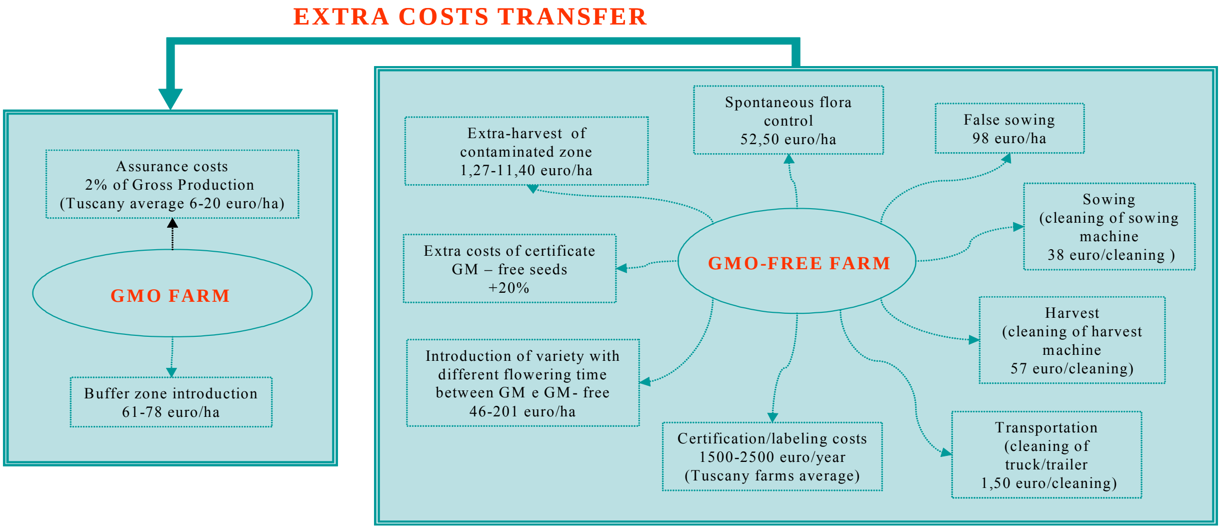


Figure 1. Costs in charge to GMO farm and GMO-free farm.

3. Scenario analysis - The combination of LCA with economic analysis (included the aspects that LCA cannot estimate, like GM toxin effect, vertical and horizontal genetic pollution, biodiversity reduction, herbicide resistant increase) it was possible to individuate three possible scenarios to describe the evolution of the agricultural system in relation with the introduction of GM crops. *First scenario* represents the current situation: no cultivation of GM crops in UE and Italy too, until the acceptance of a national and regional Coexistence Plan. In this context is active a wardship consumer system that allows the monitoring of the introduction of GM product from the foreign market on the supply chain. This scenario does not determine environmental and economic pressure, because there is not any variation from the present state of the system. *Second scenario* consists in the introduction of GM crops in controlled manner according to the Coexistence Plan. This scenario shows remarkable variations with respect to the present equilibrium both at environmental and economic level. It is characterized by a production traceability of the system also for the agricultural phase to guarantee GMO-free products without contamination. Substantially, in the second scenario it is possible to ensure the coexistence principle and the control of the environmental effects, even if the management costs increase. Finally, the *third scenario* represents a negative evolution of the situation described before, in fact it is characterized by the absence of a capable guarantee system and by an inefficiency of the respect of coexistence measures introduced, in relation with a scarce preparation and awareness of the actors involved in the chain and the repartition of the extra costs in charge to GMO-free farms. Scenario analysis allowed to individuate the three possible evolutive lines generated from GMO introduction that could allow the individuation of useful elements for the region planning in the direction of the auspice scenario that in this case is represented by the second one.

4. CONCLUSION

Coexistence scenarios resulted from this study underline the necessity of a planning management of GM crops introduction in the regional context to register environmental and economic variations and to identify the actions to undertake essential to answer at the state deviations induced in the chain context considered. In defining these aspects, an important help can come from economic analysis in combination with LCA analysis. Despite the assumptions related to the study, LCA analysis is a useful support tool in this kind of elaboration, if the availability of data concerning the studied system is ensured.

5. REFERENCES

1. Bock A K, Lheureux K, Libeau-Dulos M, Nilsagård H, Rodriguez-Cerezo E (2002) - Scenarios for coexistence of genetically modified, conventional and organic crops in European agriculture, *Technical Report Series of the Joint Research Centre of the European Commission*, EUR 20394 EN. 133 p.
2. Messéan A., Angevin F., Colbach N., Meynard J. M. (2003) - Introduction to gene flow modelling and co-existence.GMCC-03-GM Crops and Co-existence, Proceedings 13th to 14th November 2003 Denmark.
3. Messéan A., Angevin F., Gómez-Barbero M., Menrad K. and Rodríguez-Cerezo E. (2006) - New case studies on the coexistence of GM and non-GM crops in European agriculture, *Technical Report Series of the Joint Research Centre of the European Commission*, EUR 22102 EN. 112 p.
4. INRA (2004-2007) - Project SIGMEA-Sustainable introduction of GMOs into European Agriculture. Sixth framework program priority [FP6-2002-SSP1].
5. Gaugitsch H. (2002) – Experience with environmental issues in GM crop production and the likely future scenarios. Toxicology Letters 127, pp. 351-357, Elsevier.
6. Theodosiou G., Koroneus C., Moussiopoulos N. (2005) – Alternative scenarios analysis concerning different types of fuels used for the coverage of the energy requirements of a typical apartment building in Thessaloniki, Greece. Part II: life cycle analysis. Building and Environment 40, pp. 1602-1610, Elsevier.
7. http://www.gabi-software.com/
8. Bennett R., Phipps R., Strange A. and Grey P. (2004) – Environmental and human health impacts of growing genetically modified herbicide-tolerant sugar beet: a life-cycle assessment. Plant Biotechnology Journal 2, pp. 273-278, Blackwell Publisching.
9. Pimentel D. (1980) – CRC Handbook of Energy Utilization in Agriculture.

Preliminary considerations on social LCA of New Zealand dairy products

All first authors (in random order): Paula Blackett[1], Claudine Basset-Mens[1], Upananda Herath Paragahawewa[1], Bruce Small[1]

1. AgResearch Limited, Ruakura Research Centre, East Street, Private Bag 3123, Hamilton, New Zealand, Tel. +64 7 838 5213, Email. claudine.basset@agresearch.co.nz

Abstract (< 150 words)
Supply chains and products have social implications over their life cycles. Life Cycle Assessment (LCA) is internationally recognised but traditionally only considers environmental impacts. A task force set up by UNEP and SETAC defined a general framework for the integration of social aspects in LCA. However, no universal set of social indicators exist for LCA studies mainly due to the difficulty of defining a comprehensive set of indicators across the life cycle stages of a product while integrating the different socially constructed realities across countries. Case studies are needed to practically implement the framework and start developing an international database on the social performance of product life cycles. Our objective is to contribute to the early development of social LCA by trying to implement the UNEP-SETAC framework on the case study of NZ dairy products. A new team has been formed to this end involving LCA and social scientists.

1 INTRODUCTION

The concept of sustainable development was officially defined in the Bruntland report in 1987 [1]. Since then, the concept has been intuitively and widely understood but remains extremely difficult to translate into operational terms. Amongst the numerous frameworks developed to assess or implement sustainable development, the life cycle assessment (LCA) methodology has been one of the most widely recognised, namely by the United Nations [2] and EU in its integrated Policy Program [3]. The LCA methodology has been described in detail by ISO standards 14040 to 14043 and is the result of an international consensus. However, LCA traditionally only considers environmental impacts. Therefore, recommendations based on LCA fail to address possible trade-offs between environmental and social concerns. Thus, there is a need for integration of the social dimension in LCA.

2 STATE OF THE ART AND EXISTING FRAMEWORK

Over the last few years several authors have been working on integrating social dimension in the LCA methodology [4-8]. A joint task force was set up by UNEP and SETAC (Society of Environmental Toxicology and Chemistry) in 2004 on the integration of social aspects into LCA. In their feasibility study [9], different social targets and possible impact categories were defined (Fig. 1).

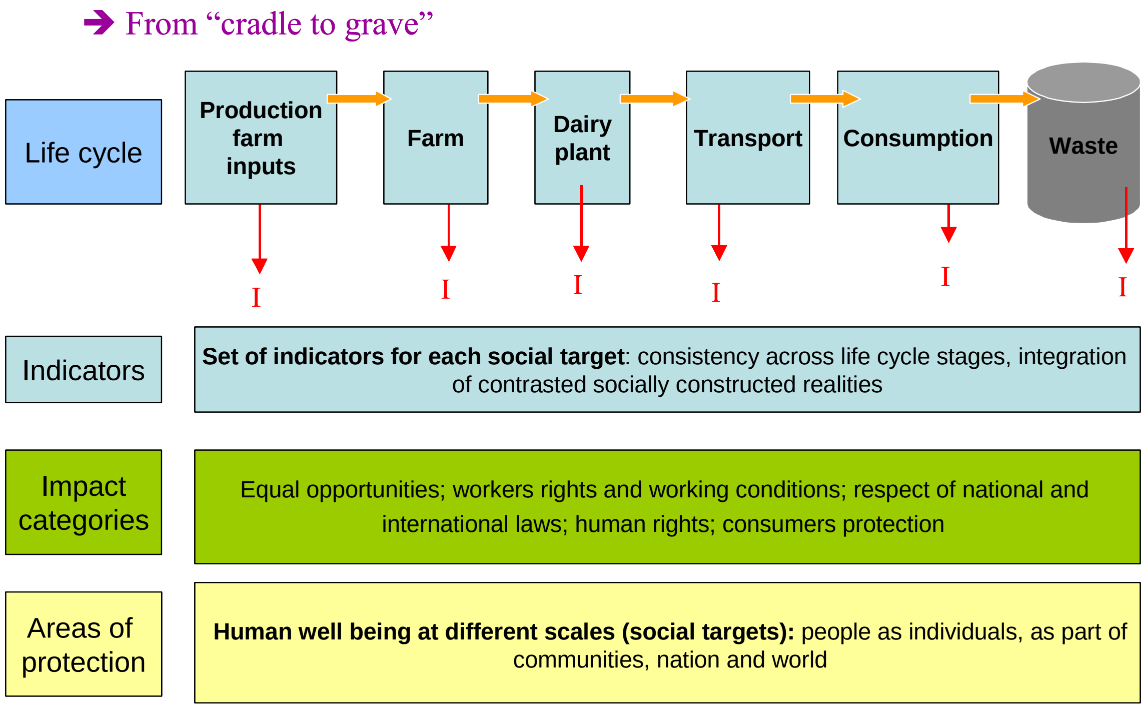


Fig. 1. General framework from UNEP-SETAC feasibility study on the integration of social aspects in the LCA methodology, adapted from [9] to NZ dairy product life cycle.

This study also highlighted the role stakeholders should play in the process. Although a broad range of frameworks with specific sets of social indicators are now available [6], they do not address social aspects of sustainability from a life cycle point of view and there is ***still no universal set of indicators to be used in social LCA studies***. Recognising the general framework proposed by [9] as a relevant basis to integrate social aspects in LCA, there are key remaining challenges and tasks for research, including:

- ⇒ Developing a set of indicators consistent across the different life cycle stages of a product taking place in various contexts across the world while integrating those different socially constructed realities
- ⇒ Preserving a practical dimension to the analysis while being comprehensive enough
- ⇒ Developing a database through a whole range of case studies all over the world
- ⇒ Preserving consistency with environmental LCA

3 OBJECTIVES

The aim of the present research project is to make a contribution to the development of social LCA by trying to implement practically the UNEP-SETAC framework for one typical case study for agriculture: NZ dairy products.

4 METHODOLOGY

LCA has been shown to be a valuable tool for the environmental evaluation of farming systems. The new challenge for LCA is to evaluate whether this framework can be used to perform a joint assessment of social and environmental impacts over the life cycle of a product.

4.1 A new team capability

To be able to address such a trans-disciplinary research question, a new team has been created at Agresearch (NZ) consisting of an LCA scientist, a social psychologist, a policy researcher and an agricultural economist who all recognised the LCA framework as very powerful for implementing their science.

4.2 Case study: NZ dairy products

4.2.1 Goal and scope

The question this research is ultimately trying to answer is whether NZ dairy products have a comparative advantage in terms of environmental and social impacts compared to their locally produced equivalent in their furthest markets (e.g. UK) [10]? The boundaries of the system have, therefore, been set up from “cradle to grave”, that is to cover the entire life cycle of dairy products produced in NZ and exported to the UK (Fig. 1). Since the purpose is to be consistent with an environmental LCA, the functional unit will be the same: one kg of cheese.

4.2.2 Inventory analysis

The core task for this part will be the identification of a consistent set of indicators. In this phase, the suitability of the work done by the social scientist members on the development of social indicators for dairy at different scales: farm, community, nation [11-13] will be analysed from an LCA point of view. The critical analysis will focus on the following points: are the proposed indicators consistent across the life cycle stages of dairy products? Do they allow integration of different socially constructed realities? Is there sufficient data available to assess those indicators? Can these social indicators be related to the functional unit as defined in LCA? We will investigate complementing the existing quantitative indicators presented in the literature with more subjective measures, in particular levels of perceived satisfaction. The value of perceived satisfaction as an indicator may rely on its consistency across countries and contexts while respecting the specific social realities. For farm and dairy manufacturing stages, the assessment will be achieved through surveys of different stakeholder populations. For upstream and downstream stages of the life cycle of NZ dairy products, the team will rely on potential published studies and on building strong networks with scientists working in corresponding countries of the life cycle, e.g. UK for consumption stage.

4.2.3 Impact assessment and interpretation

In the impact assessment stage, a joint effort between LCA and social scientists will be used to clarify the definition of inventory indicators, midpoint indicators and endpoint indicators as defined in traditional LCA methodology, but in this case, for the social dimension. It will also be important to organise and aggregate the results in a comprehensive and reduced set of indicators for the different impact categories. In the interpretation stage, the work will concentrate on analysing the robustness and the limitations of this first assessment of the social performance of NZ dairy products.

5 CONCLUSIONS AND PERSPECTIVES

A new team capability has been created on social LCA, with the objective of trying to implement the UNEP-SETAC framework through a case study for agriculture: NZ dairy products. Concentrating first on developing a consistent set of indicators at different scales, the team will be looking for networking in this new area to allow accurate coverage of dairy product life cycles and harmonised code of practice.

REFERENCES

1. Bruntland, G., Editor. 1987. Our common future: the world commission on environment and development. Oxford: Oxford University Press.
2. UNEP Division of Technology, Industry and Economics, 1999. Towards the global use of Life cycle Assessment. United Nations Publications, Paris, France.
3. Anonymous, 2003. Integrated Product Policy. Building on Environmental Life-Cycle Thinking. Communication from the commission to the Council and the European parliament commission of the European communities. Brussels, Belgium, 18.6.2003, COM(2003) 302 final.
4. Camilla Dryer, L., Hauschild, M.Z., Schierbeck, J., 2006. A framework for social life cycle impact assessment. Int J of LCA, 11(2), 88-97.
5. Labuschagne, C., Brent, A.C., 2006a. An industry perspective of the completeness and relevance of a social assessment framework for project and technology management in the manufacturing sector. Journal of Cleaner Production. Online first.
6. Labuschagne, C., Brent, A.C., 2006b. Social indicators for sustainable project and technology life cycle management in the process industry. Int J of LCA 11(1), 3-15.
7. Norris, G., 2006. Social impacts in product life cycle – Towards life cycle attribute assessment. Int J of LCA 11, (Special issue 1), 97-104.
8. Weidema, B., 2006. The integration of economic and social aspects in life cycle impact assessment. Int J of LCA, 11(special issue 1), 89-96.
9. Grießhammer, R., Benoit, C., Camilla Dreyer, L., Flysjö, A., Manhart, A., Mazijn, B., Méthot A-L., Wiedema, B., 2006. Feasibility study: integration of social aspects into LCA. Report of the task force "integration of social aspects into LCA" of the UNEP-SETAC Life cycle initiative.
10. Basset-Mens, C., McLaren, S.J., Ledgard, S., 2007. Exploring a comparative advantage for New Zealand dairy products in terms of environmental performance. LCA in Foods, 5th international conference, 25-26 April 2007, Gothenburg, Sweden.
11. Small, B., 2002. Triple bottom line product development evaluation project: Social and ethical template. Client Report for Celentis.
12. Small, B., 2006. Choices for sustainable futures: Dairy farmer preferences regarding indicators for reporting financial, environmental and social impacts at the farm scale. Client report prepared for Dairy Insight.
13. Small, B., Roth, H., Botha, N. 2007. Taumaranui Sustainable Land Management Group: Identification of Social Indicators on Farms. Client report prepared for the Sustainable Farming Fund and Meat and Wool New Zealand.

Life cycle assessment (LCA) and food miles - an energy balance for fruit imports versus home-grown apples

Michael M. Blanke
INRES- Institute of Crop Science and Resource Conservation
University of Bonn - Klein-Altendorf Research Station
Meckenheimer Str. 42
D-53359 Rheinbach
Germany, email: MMBLANKE@uni-bonn.de

Keywords: apple, energy, export, food miles, import, life cycle assessment, LCA, primary energy requirement, QS, regional production, trade, transport

Abstract. The primary energy is calculated to provide apple (cv. 'Braeburn' and 'Golden Delicious') fruit for consumers in the densely populated (8 million consumers) Rhein-Ruhr area, Germany in April. Three sources of apple fruit are compared: a) home grown-apples harvested in mid-October and CA-stored for 5 months on-site at ca. 1°C until mid March and b) fresh apples imported from New Zealand and c) fresh apples imported from South Africa in March. This was compared with apples of the same cultivar grown in a Southern hemisphere in Hawke's Bay, New Zealand or in Grabouw-Elgin, Western Cape, South Africa. These apples were picked in March with subsequent 28 day, or 14 day transport, respectively, on reefers to Antwerp for sale in April in Germany. The primary energy required for the cultivation of cv. 'Braeburn' apples in New Zealand of ca. 2.1 MJoule/kg apple fruit represented 38% of their overall primary energy requirement, compared with 2.8 MJ/kg fruit in Germany or South Africa with smaller harvests of 40 t/ha cf. 90 t/ha in New Zealand. Apples (cv. 'Braeburn' and 'Golden Delicious'), grown and stored locally in Germany, consumed nearly 6 MJ/kg fruit, which included ca. 0.8 MJoule/kg for five months CA storage during the winter. This compared favourably with 7.2-7.5 MJoule/kg for overseas shipment from New Zealand or South Africa, i.e. a 22-27% greater energy requirement for imported fruits. The CA storage of home-grown apples in Germany partially compensated for the energy required to import fresh fruit from overseas. To fully compensate for fruit imports from South Africa or New Zealand, home-grown apples had to be stored locally for ca. 9 or 18 months, respectively, i.e. in the latter case beyond the next harvest. The smaller primary energy required for domestic apple fruit is discussed with respect to providing local employment, fruit orchards preserving the countryside, fruit quality, food safety and quality assurance schemes such as QS and EUREP-GAP and food security of local fruit and networking favouring regional produce.

INTRODUCTION

Life cycle assessment calculates the primary energy , which is required to provide the consumer with a particular food item. Therein, primary energy is calculated based on the original input, e.g. coal or crude oil, and the energy which is lost through conversion e.g. into petrol, diesel or electricity. System boundaries set the limits for the processes involved therein, e.g. in this case from the fruit orchard to the home of the consumer. The accuracy and value of a LCA study improves with the widest possible system boundaries (Table 1). When comparing food chains, identical system boundaries are a pre-requisite for reliable studies (Demmeler and Burdick, 2005;

Blanke and Burdick, 2005). The energy unit of such calculations is Joule per unit food. In this contribution, Megajoule per kg fruit is employed.

Table 1: Recent misconceptions of LCA and food miles in the UK and Germany 2006

Source/ origin	Misconception	Cause of misconception	Solution-correct approach
Sunday Times 18 February 2006	Comparison of Somerset apples and NZ blueberries	Comparison of different commodities	Comparison of apples with apples
Jones (2002)	5 months storage of apples requires no energy	Electricity required for cold or CA storage	0.81 MJ/kg for 5 months storage
Schlich (2003)	Imported juice requires less energy than locally produced juice	Discrepancies in energy data, different system boundaries and calculations	Corrections by Demmeler and Burdick (2005)
Eden project New exhibit 2006	Consumer shopping requires 50% of prim. Energy	Underestimates production, storage and transport	Consumer shopping requires 1.15 MJ/kg fruit, which is 15-20% of overall primary energy (Burdick 2005)

In Germany, every other apple consumed is imported. With negligible exports, the German fruit market is very competitive with Germany being one of the world's largest fruit importers. Fruit, and particularly apple consumption, in Germany is also one of the largest in the world with 100 kg fruit and 20 kg apples/head and year. The German consumer recognises the health potential of fruit and vegetables in his diet, but is also highly critical of orchard management practises, pesticide residues and resource conservation with LCA and "food miles" only starting to become an issue, i.e. later and less of a topic than in the UK (Table 1). Recent food scares forced two German supermarkets to react and subscribe to certification and quality assurance schemes, reflecting a changing consumer attitude.

The food chain relies on a number of transport means, which greatly differ in their primary energy consumption from airfreight as the wasteful form of transport to sea cargo as the most energy efficient:

Airfreight > small truck (8 t) > medium truck (28 t) > large truck (40 t) Megatruck (80 t) > railroad > sea cargo

2 Orchard, harvest and yield data

In the present study, the primary energy is calculated to provide apple fruit in April for consumers in the densely populated Rhein-Ruhr area in Germany with ca. 8 million consumers. Three sources of apple cv. Braeburn and Golden Delicious fruit are compared: a) home grown-apples harvested in Meckenheim in mid-October and CA-stored on site for 5 months, b) fresh apples cv. Braeburn grown in New Zealand and harvested in March and c) fresh apples of cv. Golden Delicious grown in Grabouw-Elgin in the Cape region of South Africa and also harvested in March and shipped to Germany by refrigerated sea cargo (Table 2).

Table 2: Orchard data employed for the present study for food miles and LCA in 2006

Apple origin	Germany	New Zealand	South Africa
Growing area	Meckenhei m	Hawke's Bay	Grabouw-Elgin
Apple cultivar	Braeburn, GD	Braeburn	Golden Delicious
Apple acreage	M 9	MM 106	M 793/ M25

The same system boundaries are used and the same apple varieties are compared, i.e. cv. 'Braeburn' and 'Golden Delicious'. These were either grown in the Meckenheim fruit growing region in Germany, designated as home-grown, or grown in the Southern hemisphere and imported in spring into Germany.

The apple trees in Meckenheim are typically grown as slender spindles on dwarfing rootstock M9. The fruit are harvested mid October with average yields of 40 t/ha (Table 2). The apple trees in New Zealand are grown as larger trees on the more vigorous rootstock MM 106, with the New Zealand climate enabling larger yields of 90t /ha. The South African apples are grown on M793, and more recently, on M25 rootstock and produce similar yields to those in Germany.

Table 3: Harvest data for the three sources of apples

Apple origin	Germany	New Zealand	South Africa
Harvest date	Mid October	end March	mid March
Yield	40 t /ha	90 t /ha	40 t / ha
Storage	Fruit stored over winter	fresh fruit	fresh fruit

3 Fruit transport in the food chain

After harvest, the home-grown fruit are transported over ca. 10 km to the central storage facility MECO in Meckenheim with the truck returning empty. In New Zealand, the fruit are transported by truck from the Hawke's bay growing area over ca. 20 km to the coastal shipping facility at Napier on the East coast of the Northern island. In South Africa, the apples picked in March are transported from the fruit growing region in Grabouw-Elgin, Western Cape over ca. 80 km to Capetown for shipping. The fruit are cooled down before they are transported or shipped, which requires 86.3 kJ/kg. For comparative purposes, the same sized truck of 28 t is assumed which returns empty to the farm or packhouse, thereby doubling the energy required for local transport. The outward journey with the apple fruit requires cooling on the truck, while the return journey does not.

Table 4: Local transport distances (oneway) at the orchard site in the food supply chain

for home-grown versus apples imported from the Southern hemisphere

Apple origin	Germany	New Zealand	South Africa
Transport from the growing area	Meckenheim to Meco	Hawke's Bay to Napier	Grabouw-Elgin to Capetown
Transport distance	10 km	20 km	80 km
Truck	28 t	28 t	28 t

4 Energy calculation

From Napier, apple cv. Braeburn' are shipped over 23,000 km to Antwerp for distribution to the supermarkets in the Rhein-Ruhr region. Similarly, apple cv. 'Golden Delicious' fruit are shipped for 14 days over ca. 14,000 km to Antwerp for retail in the Rhein-Ruhr area in April (Table 5). Single distances are used for the cargo ships which continue their journeys rather than return to the port of origin empty loaded.

Table 5: Long distance transport (oneway) and times in the food supply chain for home-grown versus apples imported from the Southern hemisphere

Apple origin	Germany	New Zealand	South Africa
CA storage	November - March	n.a.	n.a.
	n.a.	New Zealand to Antwerp	Capetown to Antwerp
Transport distance	n.a.	23,000 km	14,000 km
Storage or transport time	5 months	28 days	14 days

n.a. not applicable

The primary energy requirement was calculated for apples (cv. 'Braeburn' and 'Golden Delicious') picked mid October in Meckenheim near Bonn, Germany with subsequent five months on-site CA storage during a Northern hemisphere winter at ca. 1°C until mid March.

Consumer shopping is based on a 3 km private vehicle driven 3 km oneway to the nearest supermarket and a purchase of 20 kg of apples a time (Kjer, 1974).

Table 6: Primary energy requirement [Mj/kg] in the food supply chain for home-grown versus apples imported from the Southern hemisphere

Source	Germany	New Zealand	South Africa
Apple production	2.80	2.10	2.80
Local transport	0.15	0.22	0.67
CA storage	0.81	n.a.	n.a.
Sea cargo	n.a.	2.83	1.41

Local distribution	0.98	1.19	1.19
Consumer shopping	1.15	1.15	1.15
Primary energy requirement [MJ/kg]	5.90	7.50	7.20

n.a. not applicable

The primary energy required for the cultivation of cv. 'Braeburn' apples in New Zealand of ca. 2.1 MJoule/kg apple fruit represented 38% of their overall primary energy requirement, compared with 2.8 MJ/kg fruit (PImentel, 1979) in Germany or South Africa with smaller harvests of 40 t/ha cf. 90 t/ha in New Zealand. Apples (cv. 'Braeburn' and 'Golden Delicious'), grown and stored locally in Germany, consumed nearly 6 MJ/kg fruit, which included ca. 0.8 MJoule/kg for five months CA storage during the winter. This compared favourably with 7.2-7.5 MJoule/kg for overseas shipment from New Zealand or South Africa, i.e. a 22-27% greater energy requirement for imported fruits. The CA storage of home-grown apples in Germany partially compensated for the energy required to import fresh fruit from overseas. To fully compensate for fruit imports from South Africa or New Zealand, home-grown apples have to be stored locally for ca. 9 or 18 months, respectively, i.e. in the latter case beyond the next harvest. The smaller primary energy required for domestic apple fruit affects employment, greenhouse gas emissions, fruit orchards preserving the countryside, fruit quality, food safety and quality assurance schemes such as QS and EUREP-GAP and food security of local fruit and networking favouring regional produce.

Conclusions

1) The larger harvests in New Zealand (90 t cf. 40 t/ha) compensate for the longer transport as sea cargo.
2) Fresh New Zealand or South African apples require ca. 25% more primary energy than home-grown (German) fruit
3) Consumer shopping alone (1.15 MJ/kg) amounts for 20-25% of the overall energy balance and is a largely under-estimated portion of the primary energy requirement.

ACKNOWLEDGEMENTS

I'm grateful to Bernhard Burdick, formerly Klimainstitut Wuppertal/ Consumer Protection Agency Düsseldorf for his input into our joint project, Ann Belcher and Peter Whitford and (ex-) staff at Bristol University, Long Ashton for information on the food miles issues and to the German Science Foundation DFG for congress grant Bl 263/10-1 to the IHC Seoul.

Literature Cited

Blanke, M.M. and B. Burdick, 2005: Food (miles) for thought – energy balance for locally-grown versus imported apple fruit. Environmental Science & Pollution Research - ESPR 12 (3), 125-127.

Carbotech, 1994: Grobabschätzung des Energieaufwandes für die Bereitstellung von ausgewählten Getränken und Nahrungsmitteln. Studie im Auftrag von Greenpeace Schweiz, Zürich, November 1994.

DEMMELER, M., BURDICK, B., 2005: Energiebilanz von regionalen Lebensmitteln – eine kritische Auseinandersetzung mit einer Studie über Fruchtsäfte und Lammfleisch. In: Kritischer Agrarbericht, ABL-Verlag, Hamm, 182-187.
JONES, A., 2002: An environmental assesssment of food supply chains: a case study on dessert apples. Environmental Management 30 (4), 560-576.
GEIER, U., FRIEBEN, B., GUTSCHE, V., KÖPCKE, U., 2001: Ökobilanz der Apfelerzeugung in Hamburg. Schriftenreihe Organischer Landbau, Köster Verlag Berlin.
KJER, 1994: Landwirtschaft und Ernährung (Studie J). In: Studienprogramm Landwirtschaft. Enquete-Kommission „Schutz der Erdatmosphäre“ des Deutschen Bundestages (Hrsg.), Economica Verlag, Bonn.
PIMENTEL, D., 1979: Food, Energy and Society, Resource and Environmental Sciences Series. Edward Arnold Publishers, London.
SCHLICH, E., 2003: Regionale Lebensmittel oft energieintensiver als “globale”. Pressemitteilung der Universität Giessen vom 4.11.2003.

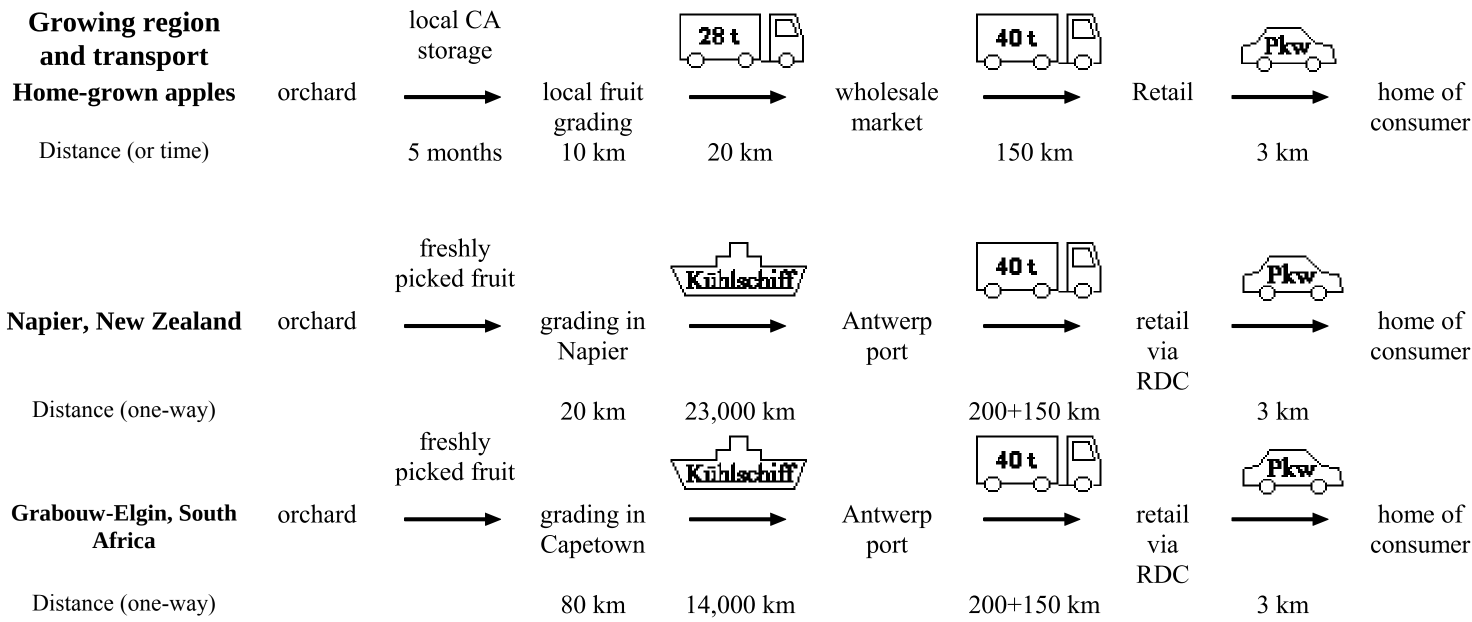


Figure 1: Transport or food chain to supply apples for consumers in Germany in April

Imported versus home-grown apples - LCA, food miles and energy balance

Michael M. Blanke [a] and Bernhard Burdick [b]

[a] Institute of Crop Science and Resource Conservation (INRES)
University of Bonn, Auf dem Hügel 6, D-53121 Bonn, Germany, email: mmblanke@uni-bonn.de
[b] Verbraucherzentrale Nordrhein-Westfalen, Mintropstr. 27, D-40215 Düsseldorf, Germany (formerly at Klima-Institut Wuppertal)

Abstract

The primary energy is calculated for imported versus home-grown apples cv. 'Braeburn' and 'Golden Delicious' for consumption in Germany in April. Southern hemisphere apples from New Zealand or South Africa are picked in March with a subsequent 28 day, or 14 day sea transport, respectively. The primary energy required for the cultivation of cv. 'Braeburn' apples in New Zealand of ca. 2.1 MJoule/kg apple fruit compared with 2.8 MJ/kg fruit in Germany or South Africa with smaller harvests of 40 t/ha cf. 90 t/ha in New Zealand. Apples cv. 'Braeburn' and 'Golden Delicious', grown and stored locally in Germany, consumed nearly 6 MJ/kg fruit, which included ca. 0.8 MJoule/kg for five months CA storage during a Northern hemisphere winter. This compared favourably with 7.2-7.5 MJoule/kg for overseas shipment from New Zealand or South Africa, i.e. a 22-27% greater energy requirement for imported fruits. The smaller primary energy required for domestic apple fruit is discussed with respect to social aspects, certification and food safety.

1. Introduction

In Germany, every other apple consumed is imported. With negligible exports, the German fruit market is very competitive with Germany and the UK being the world's largest fruit importers. Germany imports many apples from the Southern hemisphere including 65-70,000 t a year from New Zealand. Fruit, and particularly apple consumption in Germany, is also one of the largest in the world with 100 kg fruit and 20 kg apples/head and year. The German consumer recognises the health potential of fruit and vegetables in his diet, but is also highly critical of orchard management practises, pesticide residues and resource conservation with LCA and "food miles" only starting to become an issue, i.e. later and less topic than in the UK with misconceptions regarding the growing, transport, storage or handling of fruit or inconsistent system boundaries (Table 1). Recent food scares forced two German supermarkets (Aldi and Lidl) to react and subscribe to certification and quality assurance schemes, reflecting a changing consumer attitude. In the UK, two supermarkets (Marks and Spencer and Tesco) start to display the food mileage of selected items from Easter 2007.

Hence, the objective of the present study is to compare the energy required to provide customers in Germany with apples either locally-sourced or from the Southern hemisphere.

Table 1: Recent misconceptions of LCA and food miles in the UK in 2006

Source/ origin	Misconception	Cause of misconception	Solution
Sunday Times	Comparison of Somerset	Comparison of different	Comparison of apples

18 February 2006	apples and NZ blueberries	commodities	with apples
Jones (2002) [3]	5 months storage of apples requires no energy	Electricity required for cold or CA storage	0.81 MJ/kg for 5 months storage
Eden project, St. Austell, Cornwall 'Exhibit 2006'	Consumer shopping requires 50% of primary energy	Underestimates production, storage and transport	Consumer shopping requires 1.15 MJ/kg fruit, which is 15-20% of overall primary energy (Burdick 2005)

2. ORCHARD, HARVEST AND YIELD DATA

The primary energy is calculated to provide apple fruit in April for consumers in the densely populated Rhein-Ruhr area in Germany with ca. 8 million consumers. Three sources of apple cv. 'Braeburn' and 'Golden Delicious' fruit are compared: a) home grown-apples harvested in Meckenheim in mid-October and CA-stored on site for 5 months, b) fresh apples cv. 'Braeburn' grown in New Zealand and harvested in March and c) fresh apples of cv. 'Golden Delicious' grown in Grabouw-Elgin in South Africa and also harvested in March (Table 2).

Table 2: Orchard and harvest data employed for the present study for food miles and LCA

Apple origin	Germany	New Zealand	South Africa
Growing area	Meckenheim	Hawke's Bay	Grabouw-Elgin
Apple cultivar	Braeburn, GD	Braeburn	Golden Delicious
Apple acreage	M 9	MM 106	M 793/ M25
Harvest date	Mid October	end March	mid March
Fruit yield	40 t /ha	90 t /ha	40 t / ha
Storage	Oct-March	fresh fruit	fresh fruit

The same system boundaries, from the orchard to the consumer's home, are used and the same apple varieties are compared, i.e. cv. 'Braeburn' and 'Golden Delicious'. The apple trees in Meckenheim are typically grown on dwarfing rootstock M9. The fruit are harvested mid October with average yields of 40 t/ha (Table 2). The apple trees in New Zealand are grown as larger trees on the more vigorous rootstock MM 106, with the New Zealand climate enabling larger yields of 90 t /ha. The South African apples are grown on M793, and more recently, on M25 rootstock and produce similar yields to those in Germany (cf 40 t/ha).

3. FRUIT TRANSPORT IN THE FOOD CHAIN

After harvest, the home-grown fruit are transported over ca. 10 km to the central storage facility MECO in Meckenheim with the truck returning empty. In New Zealand, apple cv. Braeburn' fruit are transported by truck from Hawke's Bay over ca. 20 km to the coastal shipping facility at Napier on the East coast of the Northern island, from where they are shipped over 23,000 km to Antwerp for distribution to the supermarkets in the Rhein-Ruhr region. In South Africa, cv. 'Golden Delicious' apple fruit picked in March are transported from the fruit growing region in Grabouw-Elgin, Western Cape over ca. 80 km to Capetown for shipping for 14 days over ca. 14,000 km to Antwerp for retail in the Rhein-Ruhr area (Table 3). Single distances are used for the cargo ships, assuming that they continue their journeys rather than return empty to the port of origin. The fruit are cooled before they are transported or shipped, which requires 86.3 kJ/kg. For comparative purposes, the same sized truck of 28 t is assumed which returns empty to the farm or packhouse, thereby doubling the energy required for local transport. The outward journey with the apple fruit requires cooling on the truck, while the return journey doesn't.

Table 3: Local and long-distance transport distances (oneway) from the orchard for home-grown versus apples imported from the Southern hemisphere

Apple origin	Germany	New Zealand	South Africa
Transport from the growing area	Meckenheim to Meco	Hawke's Bay to Napier	Grabouw-Elgin to Capetown
Transport distance in 28 t truck	10 km	20 km	80 km
Sea cargo ports	n.a.	New Zealand to Antwerp	Capetown to Antwerp
Sea cargo distance	n.a.	23,000 km	14,000 km
Storage or transport time	5 months	28 days	14 days

n.a. not applicable

4 PRIMARY ENERGY CALCULATION

The primary energy requirement was calculated for apples cv. 'Braeburn' and 'Golden Delicious' picked mid October in Meckenheim near Bonn, Germany with subsequent five months on-site CA storage during a Northern hemisphere winter at ca. 1°C until mid March. Consumer shopping is based on a 3 km private vehicle driven 3 km oneway to the nearest supermarket and a purchase of 20 kg of apples at a time (Kjer, 1974) [5].

Table 4: Primary energy requirement [MJ/kg] in the food supply chain for home-grown versus apples imported from the Southern hemisphere

Source	Germany	New Zealand	South Africa
Apple production	2.80	2.10	2.80
Local transport	0.15	0.22	0.67
CA storage	0.81	n.a.	n.a.
Sea cargo	n.a.	2.83	1.41
Local distribution	0.98	1.19	1.19
Consumer shopping	1.15	1.15	1.15
Primary energy requirement [MJ/kg]	5.90	7.50	7.20

n.a. not applicable

The primary energy required for the cultivation of cv. 'Braeburn' apples in New Zealand of ca. 2.1 MJoule/kg apple fruit represented 38% of their overall primary energy requirement, compared with 2.8 MJ/kg fruit (Pimentel, 1979) [6] in Germany or South Africa with smaller harvests of 40 t/ha cf. 90 t/ha in New Zealand. Apples cv. 'Braeburn' and 'Golden Delicious', grown and stored locally in Germany, consumed nearly 6 MJ/kg fruit, which included ca. 0.8 MJoule/kg for five months CA storage during the winter. This compared favourably with 7.2-7.5 MJoule/kg for overseas shipment from New Zealand or South Africa, i.e. a 22-27% greater energy requirement for imported fruits. The CA storage of home-grown apples in Germany partially compensated for the energy required to import fresh fruit from overseas. The smaller primary energy required for domestic apple fruit affects employment, greenhouse gas emissions, fruit orchards preserving the countryside, social aspects and food safety.

5. CONCLUSIONS

1) The larger harvests in New Zealand (90 t cf. 40 t/ha) compensate for the longer sea transport.
2) Research is necessary to provide new orchard data to account for the lower input situation today- this won't change the relative outcome of the present study, but the absolute values.
3) Fresh New Zealand or South African apples require ca. 25% more primary energy than home-grown (German) fruit. This energy requirement for apple import will increase, if the bulk shipment is replaced by refrigerated containers and reefers return empty without cargo.
4) Consumer shopping alone (1.15 MJ/kg) amounts for 15-20% of the overall energy balance and is a largely under-estimated portion of the primary energy requirement.

6. ACKNOWLEDGEMENTS

We are grateful to Ann Belcher and Peter Whitford, (ex-) staff at Bristol University for local information on food miles and Bristol university for a "Visiting fellowship in Plant Physiology".

7. REFERENCES

1. M.M. Blanke and B. Burdick: Food (miles) for thought – energy balance for locally-grown versus imported apple fruit. Environmental Science & Pollution Research - ESPR 12 (3), 125-127, 2005.
2. Carbotech: Grobabschätzung des Energieaufwandes für die Bereitstellung von ausgewählten Getränken und Nahrungsmitteln. Studie im Auftrag von Greenpeace Schweiz, Zürich, November 1994.
3. A. Jones: An environmental assessment of food supply chains: a case study on dessert apples. Environmental Management 30 (4), 560-576, 2002.
4. U. Geier, B. Frieben, V. Gutsche and U. Köpcke: Ökobilanz der Apfelerzeugung in Hamburg Schriftenreihe Organischer Landbau, Köster Verlag Berlin, 2001.
5. Kjer: Landwirtschaft und Ernährung (Studie J) Studienprogramm Landwirtschaft. Enquete-Kommission „Schutz der Erdatmosphäre" des Deutschen Bundestages (Hrsg.), Economica Verlag, Bonn, Germany, 1994.
6. D. Pimentel: Food, Energy and Society, Resource and Environmental Sciences Series Edward Arnold Publishers, London, 1979.

How to prepare a less pollutant family meal?

Randi Dalgaard[a], Niels Halberg[a] and Morten Dalgas Rasmussen[b]

[a]Faculty of Agricultural Sciences, Department of Agroecology and Environment, University of Århus, P.O. Box 50, 8830 Tjele, Denmark
[b]DEG GreenTeam, Danish Agricultural Advisory Service, National Centre, Denmark.

Abstract
The environmental impacts of three typical family meals were compared using data from the Danish LCA food database. The considered impact categories were global warming, acidification and eutrophication. It was shown that substitution of pork with vegetables reduced the environmental impact, while partly substitution of conventional produced food with organic produced food was not environmentally superior using these impact categories, which do not take into account toxic effects of pesticides use.

1. BACKGROUND

Food consumption is an important contributor to global warming and eutrophication but different food items impact more than other per kg consumed as demonstrated in the literature. Generally, animal food pollutes more than vegetable food, and vegetables cultivated in heated greenhouses emit more greenhouse gasses than field-grown vegetables. There are also differences between organic and conventional products. Therefore, the consumer may, theoretically, reduce the environmental impact when composing the family meal. But, given the composite nature of meals it is not easy to estimate the relative environmental impact of substituting (part of) one food item with another.

2. OBJECTIVE

To compare the contribution to global warming, acidification and eutrophication potentials from three types of family meals with different proportions of meat and vegetables and partly substituted by organic food.

3. METHODOLOGY

The functional unit is one family meal, containing sufficient food to satisfy one family of two adults and two children. LCA data are from www.LCAfood.dk, where the EDIP method is used for the impact assessment. Consequential modelling is performed and the considered impact categories are: global warming, acidification and eutrophication. For further details on life cycle inventories and modelling of emissions from agricultural production see www.LCAfood.dk and Dalgaard et al. [1]. The LCA data on organic pork production and vegetable cultivation are based on Halberg et al. [2] and Halberg et al. [3] respectively.

4. RESULTS AND DISCUSSION

Three different kinds of family meal, containing pork, bread, milk and vegetables were defined. In table 1 the three types of family meals are shown. The first meal (standard meal) contain pork, bread, milk, bread and vegetables, and all components are conventional produced. The second meal contain less pork but more potatoes and carrots. The third meal is similar to the second meal, but pork, bread, milk, and carrots are organic.

Table 1: Composition of the three types of family meal. Org: Organic produced. Unit: kg food

Family meal:	Pork	Bread	Milk	Potatoes	Carrots	Onions	Total
1	0.8	0.5	1	0.5	0.5	0.4	3.7
2	0.4	0.5	1	0.8	0.7	0.4	3.8
3	0.4 (org)	0.5 (org)	1 (org)	0.8	0.7 (org)	0.4	3.8

The results show that a reduction of meat intake from 200 g per person to 100 g and substituting with potatoes and carrots reduce the contribution to global warming, acidification and eutrophication with 27%, 35% and 33% respectively (table 2). As shown in table 3 the environmental impact of pork is 10-20 times higher per kg product compared with the environmental impact from vegetables.

However, a comparison between family meal 2 and 3 shows that a partly substitution of conventional products with organic products increases the emissions. This is mainly due to the pork, because production of organic pork emits more greenhouse gases, acidifying and nutrifying substances compared to conventional pork [2].

Table 2: Environmental impact per functional unit

Food item	Global warming potential kg CO_2 eq.	Acidification potential g SO_2 eq.	Eutrophication potential g NO_3 eq.
1	4.6	60	333
2	3.4	39	223
3	3.8	48	235

Table 3. Comparison of specific food items used in the LCA of family meal. Only a selection of the products are presented.

Food item	Global warming potential kg CO_2 eq.	Acidification potential g SO_2 eq.	Eutrophication potential g NO_3 eq.
Pork	3.3	55	288
Low fat milk	1.2.	12	58
Bread	0.8	5	59
Potatoes	0.2	2	14
Carrots	0.1	1	4
Onion	0.4	4	15
Tomato	3.4	7	20

The study also revealed that the contribution to global warming potential is the same for pork and greenhouse cultivated tomatoes, because tomato cultivation in Danish climate demands energy for heating. Consequently a substitution of pork with tomatoes would not decrease the greenhouse gas emission. A substitution of conventional milk with organic milk would reduce the eutrophication potential, but not the global warming potential. A comparison of environmental impact from food and transport showed that the contribution to global warming potential from 'family meal 1' was equal to 14 km passenger car driving.

5. CONCLUSION

Substituting just half of the pork with potatoes and carrots reduce the contribution to global warming, acidification and eutrophication with 27%, 35% and 33% respectively. Substitution of pork with greenhouse cultivated tomatoes does not reduce the greenhouse gas emissions, but the emissions of acidifying and nutrifying substances.

If pork, bread, milk and carrots are substituted with organic products the environmental load of the family meal increases. Mainly because organic pork contributes more to global warming, acidification and eutrophication potential than conventional pork does. However, due to methodological difficulties the impact of pesticides was not considered. If it was, the environmental profiles of the organic products would obviously be improved.

6. PERSPECTIVES

Comparison of food items` environmental performance rises several questions. Is it fair to compare organic and conventional food using LCA, which presently does not adequately account for differences in pesticide emissions, animal welfare and other aspects of sustainability?

To which degree would consumers be interested in more precise information regarding the relative environmental impact if consuming different food items and choosing between organic versus conventional?

And to what extent are LCA researchers and practitioners capable of communicating our results and knowledge to consumers?

7. REFERENCES

1. R. Dalgaard, N. Halberg, I.S. Kristensen and I. Larsen
Modelling representative and coherent Danish farm types based on farm accountancy data for use in environmental assessments.
Agriculture, Ecosystems & Environment, 117: 223-237, 2006.
2. N. Halberg, J.E. Hermansen, I.S. Kristensen, J. Eriksen, N. Tvedegaard
Comparative Environmental assessment of 3 systems for organic pig production in Denmark. Submitted.
3. H. Halberg, R. Dalgaard and M.D. Rasmussen. Miljøvurdering af konventionel og økologisk avl af grøntsager. Livscyklusvurdering af produktion i væksthuse og på friland: Tomater, agurker, løg, gulerødder. Arbejdsrapport fra Miljøstyrelsen nr. 5, 2006. http://www.lcafood.dk. (Go to 'Environmental assessment of vegetables).

Does the Amazon suffer from BSE prevention?*[*]

E.V. Elferink, S. Nonhebel, A.J.M. Schoot Uiterkamp
Center for Energy and Environmental Studies IVEM
University of Groningen
Nijenborgh 4
9747 AG Groningen, the Netherlands
Telephone: + 31 50 363 71 77
Fax: + 31 50 363 71 68
E-mail: e.v.elferink@rug.nl

Abstract

In the last decade large scale production of soybeans has been a major driver of the enhanced deforestation in the Brazilian Amazon. We show that these soybeans are mainly exported to the EU to substitute for the BSE related banned meat and bone meal in livestock feed. This strongly suggests a link between Brazilian rainforest disappearance and BSE prevention.

Keywords: Deforestation, Amazon, Bovine Spongiform Encephalopathy (BSE), soybean

Deforestation of the Amazonian rainforest has been going on for more than three decades. Current Brazilian Amazon deforestation rates are the highest in the World and the highest ever in the Amazon (Laurance et al., 2002; 2001; 2004; Soares-Filho et al., 2006). This massive deforestation endangers the vital roles of the Brazilian Amazon in ecological and environmental key processes such as maintaining biodiversity and terrestrial carbon storage (Laurance et al., 2001; Fearnside, 1997; Schaeffer and Rodriques, 2005). To stop this process insight in the drivers of deforestation is essential. The drivers are complex and involve an interaction of cultural, demographic, economic, technological, political and institutional issues (Schaeffer and Rodriques, 2005). Yet, rainforest is mainly cleared for the conversion of land for agricultural purposes such as industrial soybean farming (Fearnside, 2001). The latter is seen as one of the key economic and political reasons behind deforestation for agricultural purposes (Laurance et al., 2004). In the last decade industrial soybean farming doubled its area to $22*10^6$ ha now being the largest arable land user in Brazil (Figure 1). A doubling made possible by a new variety of Brazilian soybeans which flourish in the Amazonian climate (Mongabay, 2006) but also by Brazilian national programs like "Avança Brasil" and its predecessor "Brasil em Ação" making the enormous potential of agricultural land in the Amazon accessible (Fearnside, 2002).

The demand for soybeans is subjected to global market forces for soybeans, oil and scrap. A small quantity is consumed directly by humans but the bulk is used for the derivatives oil and scrap. The most important driver of soybean production is the use of soybean scrap as a feed source (Weidema, 2000). Soybean scrap is a high quality protein source ideally suited for the increasing protein needs of modern day livestock.

[*] Published in: Agriculture, Ecosystems and Environment 120 (2007) 467-9

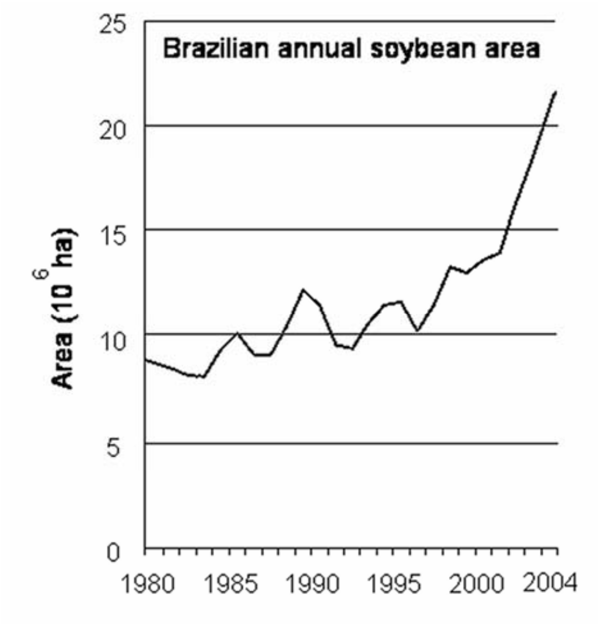


Figure 1 Area under cultivation for soybeans in Brazil since 1980. Data are from the FAOSTAT agricultural database.

The relation between deforestation and soybean production can also be observed in export statistics (Figure 2). Brazilian export of soybeans has doubled in the last ten years mainly due to increased exports to the European Union (EU) and China. The increased need for soybeans in China can be ascribed to increased livestock product consumption in China. However, this is not the case for the EU. Statistics show that livestock production in the EU remained constant over the last decade (FAOSTAT, 2006). This indicates that soybeans are used to replace another livestock feed. With respect to feed ingredients one major event stands out in the last decade: the ban on using animal by-products in feed as a result of the BSE-affair.

Figure 2 Brazilian production and export of soybean equivalents since 1980. Data are from Oil World ISTA Mielke GmbH.

In 1995 the first human deceased from variant Creutzfeld-Jacob disease as a result of eating infected beef (Will and Ironside, 1996). To stop this health threat more stringent regulations were imposed in the EU. They included a ban on meat and bone meal (MBM) in feed for ruminants. The feed ban was later extended to a ban on feeding processed animal based proteins to all farmed animals which are kept, fattened or bred for the production of food (European Communities, 2006). Before the BSE-affair 10% of the feed originated from MBM. Nowadays MBM is combusted as biomass to yield CO_2-neutral energy. MBM, a high quality protein component for feed, is largely replaced by soybean scrap. Due to BSE regulation the EU replaced $16*10^6$ t MBM feed equivalent to $23*10^6$ t soybeans (European Communities, 2002). With an average soybean yield of 2.3 t/ha an area of $10*10^6$ ha, corresponding to 10% of the EU arable land, is required. In reality the EU produces hardly any soybeans but it is the largest consumer of soybean scrap in the world. Over the last ten years the EU annual soybean import for feed increased with $17*10^6$ t (Ista Mielke GmbH, 2005). An equivalent amount is needed to compensate for the MBM loss. The EU soybeans are primarily imported from Brazil as a result of the EU position towards GMO's (Schofield, 2002). Brazil is the only large soybean producing country that officially prohibits the cultivation of GM soybeans. In contrast, China the worlds fastest growing soybean consumer imports its soybeans mainly from the USA the world largest producer of soybeans (Ista Mielke GmbH, 2005).
The EU BSE regulation combined with its attitude towards GMO's resulted in an increased demand for soybean scrap of Brazilian origin. The large area required to cultivate the enhanced EU soybean demand is of the same size as the area deforested in Brazil since 1996. This strongly suggests that the Brazilian rainforest is suffering from BSE prevention. However, the effect of the European livestock system on rainforest depletion in Brazil is rather complex. Several interesting aspects as the consequences of using alternative MBM substitutes or even continued use of MBM as well as what will happen in Brazil if EU would stop importing soybean scrap are outside the scope of our study but they deserve further research.

References

European Communities, 2002. The use of animal proteins in the feed of farmed animals.
http://ec.europa.eu/food/index_en.htm
European Communities, 2006. Community legislation on BSE.
http://ec.europa.eu/food/index_en.htm
FAOSTAT, 2006. Agricultural data, last accessed April 2006.
Fearnside, P.M., 1997. Greenhouse gases from deforestation in Brazilian Amazonia: net
comitted emissions. Climatic Change, 35, 321-360.
Fearnside, P.M., 2001. Soybean cultivation as a threat to the environment. Environmental
Conservation, 28, 23-38.
Fearnside, P.M., 2002. Avanca Brasil: Environmental and Social Consequences of Brazil's
Planned Infrastructure in Amazonia. Environmental Management, 30, 735-747.
Ista Mielke GmbH, 2005. Oil World. Hamburg.

Laurance, W.F., Albernaz, A.K.M., Fearnside, P.M., Vasconcelos, H.L., Ferreira, L.V., 2004. Deforestation in Amazonia. Science, 304,1109-1111.

Laurance, W.F., Albernaz, A.K.M., Schroth, G.T., Fearnside, P.M., Bergen, S., Venticinque, E.M., Da Costa, C., 2002. Predictors of deforestation in the Brazilian Amazon. Journal of Biogeography, 29, 737-748

Laurance, W.F., Cochrane, M.A., Bergen, S., Fearnside, P.M., Delamonica, P., Barber, C., D'Angelo, S., Fernandes, T., 2001. The future of the Brazilian Amazon. Science, 291, 438-439.

Mongabay, 2006. Amazon soy becomes greener. http://news.mongabay.com/2006/0725-amazon.html

Schaeffer, R., Rodriques, R.L.V., 2005. Underlying causes of deforestation. Science, 307, 1046-1047.

Schofield, G., 2002. EU regulation of genetically modified organisms: Food and feed, traceability and labelling. Journal of Commercial Biotechnology, 9, 27.

Soares-Filho, B.S., Nepstad, D.C., Curran, L.M., Cerqueira, G.C., Garcia, R.A., Ramos, C.A., Voll, E., McDonald, A., Lefebvre, P., Schlesinger, P., 2006. Modelling conservation in the Amazon basin. Nature, 440, 520-523.

Weidema, B., 2000. Avoiding Co-Product Allocation in Life-Cycle Assessment. Journal of Industrial Ecology, 4, 11.

Will, R.G., Ironside, J.W.,1996. A new variant of Creutzfeldt-Jakob disease in the UK. Lancet, 347, 921.

Comparative LCA of liquid milk: pasteurised versus sterilised milk

A. Hospido; M.T. Moreira; G. Feijoo

Department of Chemical Engineering. School of Engineering.
University of Santiago de Compostela, 15782 – Santiago de Compostela (Spain).

Abstract
Galicia (NW Spain) is by far the main producer of milk in Spain; however, this outstanding position is not further maintained at the processing level and around one quarter of the produced milk leaves the region for further processing elsewhere.
Among the diverse liquid milk available at the market, sterilised milk is the most demanded option, representing more than 90% of the consumption. However, several companies have recently showed an interest in promoting the consumption of pasteurised milk.
This paper presents some preliminary results on the comparison of pasteurised and sterilised milk.

1. INTRODUCTION

As in European Union, food industry in Spain occupies an outstanding position with more than 17% of the total turnover of final products and more than 20% of the total expense of raw materials [1]. In particular, dairy sector in Galicia (NW Spain) is very significant as it is shown in Table 1, which displays the contribution of regions and activities.

Table 1: Sorting by region and activity, adapted from [1]

Classification by Autonomous Community		Classification by Sub-sector of Activity	
Region	Contribution (%)	Sub-sector	Contribution (%)
Andalucia	18.5	Bread, baking and biscuits	41.5
Catalonia	11.9	Meat processing	13.8
Castilla - Leon	10.0	Wine	12.5
Castilla – La Mancha	8.5	Fats and oils	5.2
Galicia	8.2	Dairy products	5.1

Under a consumption perspective, liquid milk is by far the most consumed product. Among the diverse types of milk available in the market, its distribution has changed and nowadays dominance of sterilised milk is clear at Spanish households (Table 2).

Table 2: Milk consumption at Spanish households [2, 3]

%	1987	1997	2005
According to thermal treatment			
Sterilised	53	89	95
Pasteurised	19	4	2
Raw	28	7	3

Some companies have this year promoted a campaign on consumption of pasteurised milk, focusing their pressure on product freshness. Pasteurised milk is also the most demanded product in many European countries. This paper presents preliminary results of a comparative LCA between the two types of processed milk.

2. SYSTEM UNDER STUDY

Liquid milk has extensively been studied under a life cycle perspective [4-9]. Figure 1 shows the life cycle stages that are normally considered in those studies:

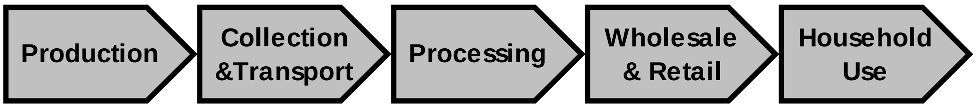


Figure 1: Subsystems involved at the life cycle of milk

The system function has been defined as the consumption of 1 L of packed milk at home. No distinction has been made on milk fat content.

3. INVENTORY DATA

As already mentioned, several LCA of milk production and processing are available elsewhere; to the best of our knowledge, the majority of those studies have dealt with pasteurised milk [4-7]. Only Hospido et al. [8, 9] evaluated the production of sterilised milk. Among them, two have been selected for this preliminary comparison:

- Swedish scenario for pasteurised milk [7]
- Galician scenario for sterilised milk [9]

4. RESULTS

Table 3 shows a direct comparison concerning the contribution to several impact categories[1] of the different steps along the milk chain, in order to find similarities and dissimilarities:

Table 3: Relative contribution per stage to the entire life cycle of milk

	Production at Farms	Raw Milk Collection	Processing at Dairies	Milk Distribution	Market and Consumer
			Pasteurised Milk – Sweden [7]		
EU	98.2	0.20	0.93	0.51	0.16
AC	97.4	0.38	0.84	0.95	0.42
GW	87.0	0.75	5.66	1.87	4.71
			UHT Milk – Galicia (Spain) [9]		
EU	94.74	2.03	1.00	2.17	0.08
AC	84.07	4.49	5.79	4.83	0.82
GW	72.51	6.00	11.86	6.45	3.18

- Raw milk production: Not surprisingly, production of milk at dairy farms is by far the major source in all impact categories in both scenarios.
- Milk processing: The contribution of milk processing is higher when sterilised milk is considered, with two elements as the main contributors:

a) After pasteurisation, milk is sterilised by Ultra High Temperature (UHT) treatment (137–140°C for 3-4 s). This extra step involves an important demand of energy and cleaning agents (26% of electricity, 50% of thermal energy, 57% of NaOH (CIP units) and 52% of HNO_3 (CIP units) are used at the sterilisation step).

[1] EU = eutrophication, AC = acidification, GW = global warming

b) Sterilised milk requires packaging under aseptic conditions, so it can be stored at room temperature for months. So far, the most widespread material for this application is the Tetra Brick Aseptic package developed by TetraPack; however, manufacture of the packaging material stands for 27% of the electricity consumed at dairies [9]. Table 4 compares two brick-shaped containers for milk (tetrabrik for sterilised and cardboard box for pasteurised) and shows that, from an environmental point of view, the aseptic container seems to be a poor alternative.

Table 4: Environmental comparison of carton containers for market milk (1 L)

	Tetra Brick [10, 11]	Carton Container [7]
GW (g CO_2)	74.09	37.90
AC (g SO_2)	0.51	0.10
EC (MJ)	2.23	1.51

- Transportation: The input of transports, both collection of raw milk and distribution of final product, is more important in the Galician scenario. Although the Swedish report did not provide information regarding average distances, they could be considered similar to those reported in another report [5]: 70 km for collection and 58 km for distribution. These figures are much lower than the numbers reported at [9]: 296 km for collection and 484 km for distribution. Nevertheless, pasteurised milk would require refrigerated trucks for its distribution, while sterilised milk does not, which entails an extra consumption of fuel and therefore, higher environmental burdens per km.
- Retailers and consumers: The differences detected at this stage are connected to certain extend to packaging characteristics. At the Swedish scenario, extra energy is required for refrigerated storage, which is unnecessary if UHT process is performed.

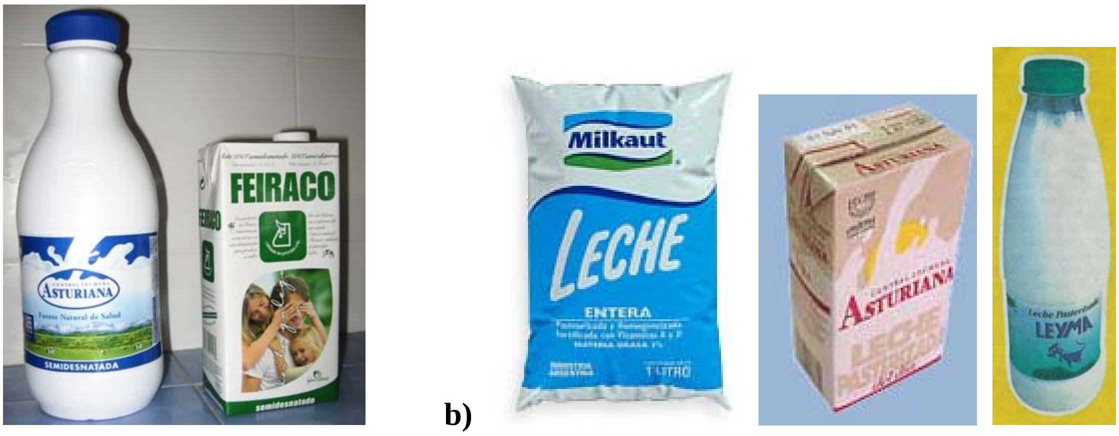
5. FUTURE OUTLOOK

The results here presented are preliminary and, as scenarios are not directly comparable, no absolute values have been presented. The Galician scenario is changing nowadays and, as introduced above, some dairies are promoting consumption of pasteurised milk. We are now shaping the study according to the following research questions:

a) Is pasteurised milk a less-environmental damaging option than sterilised milk?
b) Can the extra step at processing and the most resources demanding packaging of sterilised milk be balanced with the extra energy that pasteurised milk requires along the chain?
c) Among the alternatives available at the market (Figure 2), which is the best option for pasteurised milk and which for sterilised milk?

A priori, the system function will be measured as the consumption of 87 L of milk[2], packed in the correspondent format (that means 87 containers of 1 litre or 58 containers of 1-5 litres), in a Galician household. No distinction has been made on the milk fat content.

[2] Annual consumption per capita in 2005 [3]



a) b)

Figure 2: Milk containers. a) Sterilised milk (1.5 L plastic bottle and 1 L tetrabrick), b) Pasteurised milk (1 L plastic bag, 1 L carton brick and 1 L plastic bottle)

6. ACKNOWLEDGEMENTS

This work has been financed by Xunta de Galicia (Ref: PGIDIT04TAL262003PR) and Almudena Hospido is funded by the *Isidro Parga Pondal* programme.

7. REFERENCES

[1] MAPA (2004): Industria Agroalimentaria [In Spanish]. Available at: http://www.mapa.es/ministerio/pags/hechoscifras/espanol/pdf/17.pdf (Last accessed 16/04/2007)

[2] C. Fuente (2003): Consumos en España (año 2001): Leche y Derivados [In Spanish]. Seminario de la Asociación Española de la Economía Agraria, Lugo (Spain). Available at: http://www.usc.es/~idega/mesaredonda.html (Last accessed 16/04/2007)

[3] MAPA (2005): La alimentación en España 2005 [In Spanish]. Available at: http://www.mapa.es/alimentacion/pags/consumo/2005/panel-05.pdf (Last accessed 16/04/2007)

[4] M.H. Eide and T. Ohlsson (1998): A comparison of two different approaches to inventory analysis of dairies. International Journal LCA 3(4):209-215.

[5] M.H. Eide (2002): Life Cycle Assessment of Industrial Milk Production. International Journal LCA 7(2):115-126.

[6] P.H. Nielsen (2003): Milk production (LCA Food database - Denmark). Available at: http://www.lcafood.dk/processes/industry/milkproduction.html (Last accessed 16/04/2007)

[7] Svenskmjölk (2003). http://www.svenskmjolk.se/pdf/Milk_and_the_Environment_booklet.pdf. Milk and the Environment. (Last accessed 16/04/2007)

[8] A. Hospido, M.T. Moreira and G. Feijoo (2003): Simplified life cycle assessment of Galician milk production. International Dairy Journal 13: 783–796

[9] A. Hospido (2005): Life Cycle Assessment as a tool for analysing the environmental performance of key food sectors in Galicia (Spain): milk and canned tuna. Doctoral Thesis, Department of Chemical Engineering, University of Santiago de Compostela (Spain).

[10] TetraPak-España (2000). Declaración Ambiental [In Spanish]. Madrid.

[11] TetraPak-España (2001). Informe de Sostenibilidad 2000 [In Spanish]. Madrid.

Environmental impacts of Finnish greenhouse cucumber production systems

J.-M. Katajajuuri[a], J. Grönroos[b], A. Mikkola[a], J. Näkkilä[c] & P. Voutilainen[a]

[a]MTT Agrifood Research Finland, Biotechnology and Food Research, Food Ecology, FI-31600 Jokioinen
[b]Finnish Environment Institute. Research Programme for Production and Consumption
P.O. BOX 140, FI-00251 Helsinki, Finland
[c]MTT Agrifood Research Finland, Plant Production Research, Horticulture, Toivonlinnantie 518, FI-21500 Piikkiö

Abstract
Life cycle assessment was conducted for three Finnish greenhouse cucumber production systems: seasonal cultivation, traditional cultivation and year-round cultivation, with the typical cultivation periods of 4, 8 and 12 months per year, respectively. The results were dominated by the cultivation phase. The energy in cucumber product system was mainly used for heating of the greenhouses and – in the case of year-round cultivation – for artificial lighting of the cucumber plantations. From the energy use and climate change perspective year-round production caused more environmental burdens compared to traditional and seasonal production. However, the traditional production caused more acidifying emissions than the others. The eutrophying emissions during the life cycle of cucumber originated mostly from the nutrients that have drifted out from the greenhouses due to the irrigation surplus. Due to the lack of reliable data the emissions affecting eutrophication were assumed to be equal in all production systems.

1. INTRODUCTION

The total production volume of the Finnish greenhouse cucumber was 31 thousand tonnes in 2004. The total production acreage was 75.6 hectares of which circa 20 hectares were year-round cultivation. The average cucumber yield was 41 kg/m^2. However, the m^2-yields differ remarkably between different production systems.

The aim of the study was to increase the knowledge on the environmental burdens of the Finnish cucumber production chain: to find out the contribution of the different production phases on the emissions and energy consumption in the system, and to recognize the potential measures to improve the environmental performance of Finnish cucumber production. The data collection and respective data quality assessment was especially concentrated on year-round production due to the fact that circa half of the cucumber production volume is taken place as year-round cultivation in Finland.

2. MATERIALS AND METHODS

Three different greenhouse cucumber cultivation systems were examined [1]:

1. Seasonal cultivation:	4-6 months cultivation time in natural sun light conditions
(short period cultivation)	annual yield 11-35 (average 23) kg m^{-2} from one cucumber planting
	typically renewal umbrella training system
2. Traditional production:	7-8 months cultivation time in natural sun light conditions or with low capacity supplemental lighting
	annual yield 29-43 (35) kg m^{-2} from two cucumber plantings
	typically renewal umbrella training system

3. Year-round production high capacity supplemental lighting enables year-round production annual yield 71-115 (84) kg m^{-2} from tree or four plantings typically high wire training system

The overall product system of the Finnish greenhouse cucumber (incl. seedling stage) and the main environmental burdens related to it was under examination. The aim was to include all data on the use of energy and natural resources and the emissions related to them and to the cucumber cultivation itself.

The production of fertilizers, growing media, packaging, fuels and electricity were included to the product system as well as the major transport systems of raw materials and the final products. Production of cucumber seeds, chemical and biological control agents were excluded from the product system because of the lack of data. Impacts on landscape and possible problems with noise and artificial lighting were omitted as well.

The functional unit (FU) was 1000 kg greenhouse cucumber (including the quality classes I and II) bought by consumers from retail shops. Besides data on cultivation measures were collected directly from seven producers, also Horticultural Enterprise Register and Finnish market garden association were used as information sources. The most accurate data of cultivation inputs and outputs were got from year-round production greenhouses. Data for the other life cycle stages were acquired mainly directly from the manufacturing industry and logistic companies. Cucumbers were modelled to be centrally delivered over Finland according to the current regional market shares.

The main impact categories were climate change, aquatic eutrophication and acidification. In Characterization the site-dependent characterization factors were used for the two latter impact categories [2,3]. For climate change the IPCC 2001 [4] factors were used.

3. RESULTS

The results were dominated by the cultivation phase. Primary energy demand per FU for year-round production system was three times higher compared to seasonal production and twice as much as traditional cultivation due to the total energy demand of cultivation. The energy is especially used in heating of the greenhouses and – in the case of year-round cultivation – artificial lighting of the cucumber plantations. Most of the primary energy sources were non-renewable ones. Climate change potentials between systems were also explained by the energy use in greenhouses. Year-round cultivation produced climate change potential twice as much per FU as seasonal cultivation, and 30% more than traditional cultivation.

From the production systems studied, traditional cultivation generated more emissions causing acidification than the other production systems. This was due to the fact that in traditional cultivation were used more heavy oils than the other systems studied. Even though the primary energy use in year-round cultivation was much higher than in traditional system, the intensive use of electricity in year-round cultivation and the lower acidification potential of electric power compared to the use of oil-based fuels explain the result.

The emissions causing eutrophication during the life cycle of the cucumber originated almost totally from the nutrients that have drifted out from the greenhouses due to the irrigation surplus. The nutrient run-off potential assessment was based on average irrigation surplus percentage (15% for peat and 22% for rockwool and perlite) and the data of fertilization levels was obtained from the growers. Furthermore, approximately 30% of irrigation surplus of greenhouses in Finland are collected and treated or recycled [5]. The Finland-specific transportation factors related to the nutrient emissions to the waters (75% for N and 100% for P) were included in the LCIA [2]. It also was supposed that both nitrogen and phosphorous from greenhouses were totally in algal-available form. Based on the above mentioned parameters, the emissions of eutrophicating nutrients for year-round production was estimated to be around 0,45 kg N/1000 kg and 0,13 kg P/1000 kg cucumber. Due to the lack of reliable data on fertilization and respective irrigation surplus it was not possible to assess nutrient emissions for seasonal and traditional cultivation. Thus, emissions affecting eutrophication were assumed to be equal in all production systems per FU.

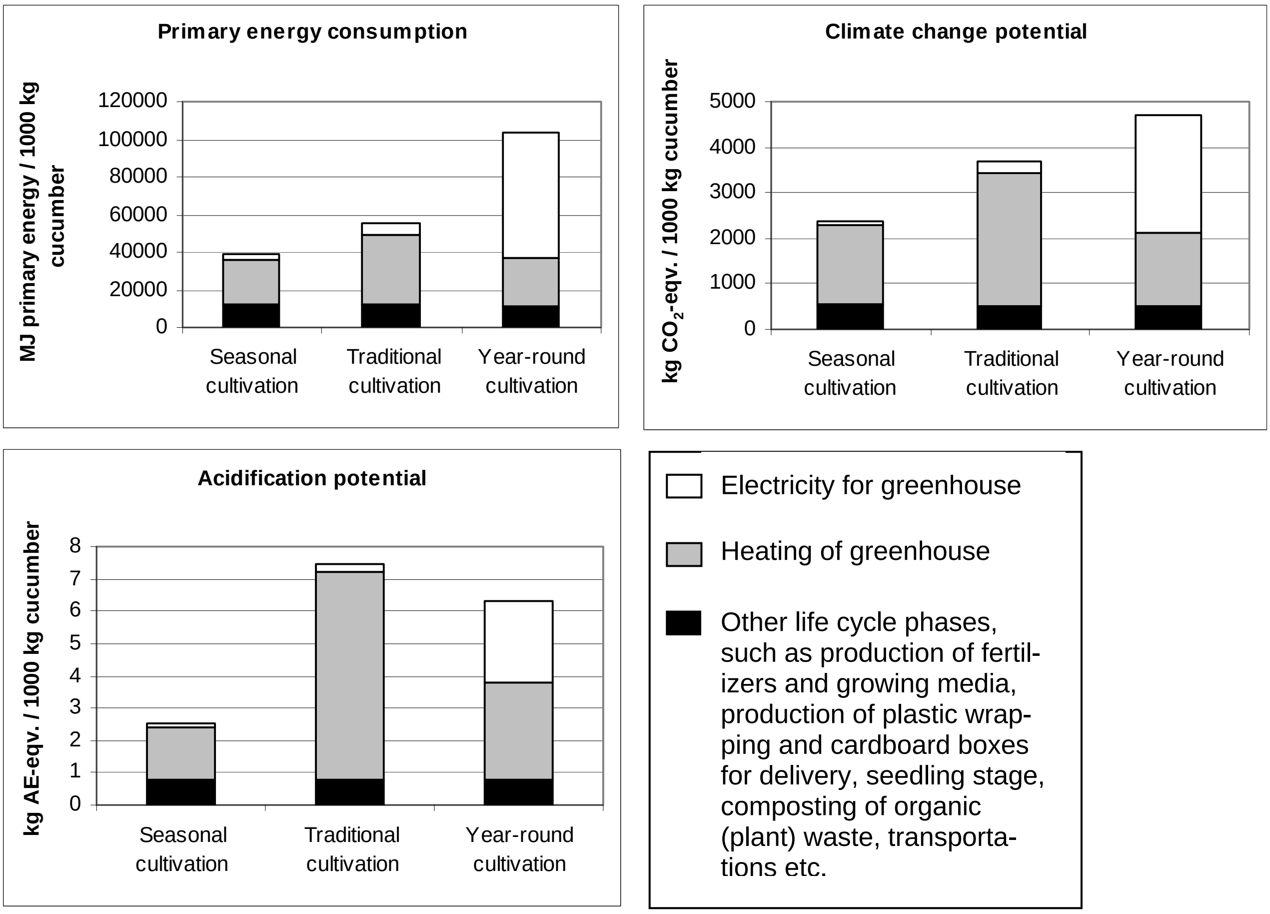


Figure 1. Primary energy demand, climate change impact and acidification impact of cucumber production systems per 1000 kg cucumber.

4. DISCUSSION

From the energy use and climate change perspective year-round production caused more environmental burdens compared to traditional and seasonal production. However, the traditional production caused more acidifying emissions than the others. Nutrient releases could be a bit lower in year-round production than in other production systems, but due to the lack of reliable data the emissions affecting eutrophication were supposed to be equal in all production systems.

There were some differences in cultivation practices between the year-round greenhouses, especially in the use of heat per FU. Relatively large use of electricity was compensated up to a point by lower use of heat. However, this does not explain the overall difference, because when comparing the total energy (kWh heat and electricity) use per FU, the difference between the largest and lowest energy user was 27%. Besides the energy use there were differences in yields between the year-round greenhouse enterprises. Any specific explanation was not found for the differences due to the fact that for example energy use is a sum of many factors, such as roofing and ceiling materials used in greenhouses, cultivation method, use of thermal screens, tightness of greenhouse structures, and geographical location and position of the greenhouse.

Contrary to other countries, because Baltic Sea and inland waters are sensitive to nutrient releases, also eutrophication is important impact category in Finland. This can be seen when the total environmental impacts of cucumber production were evaluated and demonstrated using the Finnish Eco-Benchmark [6]. For the year-round production, the total environmental impacts of the greenhouse cucumber production were for around 30% due to the climate change and for 20% due to eutrophication, acidification and primary energy consumption for each. Furthermore, according to the Eco-Benchmark comparison, the daily consumption of cheese (30 g per person per day) created ten times as much environmental impacts as the daily consumption of cucumber (22 g), and the daily consumption of rye bread (83 g) twice as cucumber, respectively. Using only ‘primary energy Eco-Benchmark’, the daily consumption of cucumber would count about 80% of the energy use of cheese consumption, and cucumber production would need 140% of energy use of rye bread production. This clearly highlights the energy intensiveness of greenhouses in Finland, which differs a lot compared for example to the greenhouses in Mediterranean are (see e.g. [7, 8]).

The emissions causing eutrophication in cucumber system originated mostly from the nutrients that have drifted out from the greenhouses to the environment due to the irrigation surplus. However, these emissions could be largely decreased by recycling of the irrigation water, using site-specific treatment systems for the wastewaters or by leading the wastewaters into the municipal wastewater treatment plant. As expressed per unit of cultivation area, the nutrient emissions from greenhouse production are multiple compared to the crop production, if no emission control measures are applied. Therefore, even the total acreage and, thus, the total emissions of greenhouse production are much lower than those of agriculture, greenhouse production may have a significant impact for example on eutrophication or on the state of ground waters on the local scale. Because of the point-source nature of the nutrient emissions of the greenhouses, the emissions can be controlled more easily than the emissions from the field cultivation. The emission controlling measures are the same as mentioned above.

5. REFERENCES

1. J.-M Katajajuuri, A. Mikkola, J. Grönroos, P. Voutilainen, J. Näkkilä and T. Hovi-Pekkanen
   The environmental impacts and related improvement possibilities of greenhouse cultivated cucumber in Finland. (In Finnish.) Maa- ja elintarviketalous. 55 p. 2007. In Press.
2. J. Seppälä, S. Knuuttila and K. Silvo
   Eutrophication of aquatic ecosystems. A new method for calculating the potential contributions of nitrogen and phosphorus.
   International Journal of LCA 9 (2): 90–100, 2004.
3. J. Seppälä, M. Posch, M. Johansson and J.-P. Hettelingh
   Country-dependent characterisation factors for acidification and terrestrial eutrophication based on accumulated exceedance as an impact category indicator.
   International Journal of LCA 11 (6): 403-416, 2006.
4. J. Houghton, Y. Ding, D. Griggs, M., Noguer, P. van der Linden, X. Dai, K. Maskell and C. Johnson, C. (eds.)
   IPPC Climate Change 2001: The Scientific Basis. Contribution of working group I to the third assessment report of IPCC. 786 s. (http://www.grida.no/climate/ipcc_tar/wg1)
5. J. Grönroos and A. Nikander
   Greenhouse production and the environment. Results of the query. (In Finnish.) Mimeograph series of the Finnish Environment Institute 257. Helsinki. 45 p. 2002.
6. A. Nissinen, J. Grönroos, E. Heiskanen, A. Honkanen, J.-M. Katajajuuri, S. Kurppa, T. Mäkinen, I. Mäenpää, J. Seppälä, P. Timonen, K. Usva, Y. Virtanen, and P. Voutilainen

Developing benchmarks for consumer-oriented life cycle assesment-based environmental information on products, services and consumption patterns. Journal of Cleaner Production 15 (6): 538-549, 2007.

7. A. Carlsson-Kanyama
   Food Consumption Patterns and their Influence on Climate Change. Greenhouse Gas Emissions in the Life-cycle of Tomatoes and Carrots Consumed in Sweden. Consumption Patterns and Climate Change: Consequences of eating and traveling in Sweden. Doctoral thesis in Natural Resources Management. Stockholm. 1999.
8. M.A. Antón, F. Castells, J.I. Montero and P. Muñoz
   Most significant substances of LCA to Mediterranean Greenhouse Horticulture. Life Cycle Assessment in the Agri-food Sector. 4th International Conference, October 6-8, 2003, Denmark. P. 199-204. Conference proceedings.

ACKNOWLEDGEMENTS

This case study was a part of Finnish *Foodchain (2000-2007) LCA studies.* We wish to acknowledge The Finnish Environmental Cluster Research Programme for the financial support.

Environmental impacts of product packaging in Finnish food production chains

J.-M. Katajajuuri[a] and Y. Virtanen[a]

[a]MTT Agrifood Research Finland, Biotechnology and Food Research, Food Ecology, FI-31600 Jokioinen

Abstract

Product packaging has been the most common target of regulation aimed at reducing the environmental effects caused by consumption. This paper discusses the contribution of packaging to the environmental impact of food production chains. The results obtained in LCA studies on a number of Finnish food production chains indicated that the contribution of packaging to the total environmental impact, expressed in the Finnish Eco-Benchmark scale, is less than 2% for most food products. The order of magnitude was found about the same for specific environmental impacts. For beer, that includes much water, the relative importance of packaging was found substantially higher than for products that contain little water. The most of the environmental impacts were found to rise from other activities than packaging. Packaging prevents unnecessary product losses in logistic chains, retail and households, and thus helps to reduce the environmental impact of the whole of the food supply chain.

1. INTRODUCTION

Product packaging has been the most common target of LCA studies and regulation aimed at reducing the environmental impacts caused by consumption. Packaging has come to symbolize the issue of waste, but it is increasingly taken up as a key issue also in discussions on other environmental concerns by the media, environmentalists and consumers.

Food products differ from other products for the functional requirements and importance of packaging. In practise, packaging is a necessity to protect the safety and the quality of food products. High standards for safety and quality set special requirements for food packaging in order to ensure high safety and quality throughout the whole logistic chain, from production sites via retail to households and end-use. Packaging should also support the avoidance of unnecessary food losses in the logistic chain. The special kinds of functional requirements increase the interest in the question of the environmental impacts of packaging of food products.

We discuss in this paper the contribution of packaging to the environmental impact of food production chains on the basis of the results obtained in LCA studies on a number of Finnish food production chains. We will find out that the contribution of packaging to the total environmental impact of a food product chain is relatively small. However, there are differences between the packaging systems. Therefore, it is important to assess the product systems as a whole, including the trade-off effects between the environmental loads caused on one hand by lost food and on the hand by the changes in the packaging systems.

2. MATERIALS AND METHODS

The food products included in the LCA studies referred to in this paper were all Finnish. They addressed broiler chicken meat, frozen potato gratin, oatmeal, hard cheese, potato flour, lager beer and greenhouse cucumber. The main objective of these recent LCA studies was to compile reliable environmental impact data for all phases of these food product systems, from the production of farm inputs to retail stores. Consumer shopping trips were not included, but in two case studies cooking was included. Data for the system models, including packaging production, were mainly acquired from the manufacturing industry, thus providing a reliable basis to analyse the sources of environmental impacts. The studies were carried out between 2002 and 2007 [1,2,3,4]. The packaging studied included primary, secondary and tertiary packaging, and their material production chains. For certain basic packaging materials data from APME (plastics), FEFCO (paper board) and EAA (aluminium) was used. Conversion data to packaging, for example plastic bottles, films and multilayer materials and cardboard boxes, were mainly collected from the producers [see e.g. 5].

The impact categories included in each study were primary energy consumption, climate change, acidification, formation of tropospheric ozone, and aquatic eutrophication. Additionally, in the three most recent studies (broiler chicken meat, lager beer and cucumber) an indicator was calculated for the total environmental impact. This indicator, called the Eco-Benchmark value, is based on the daily per capita consumption of the studied product. In order to obtain the Eco-Benchmark value life-cycle impacts are assessed for the daily consumption, then normalised using the per capita impact rates of Finland for each impact category, and finally summed up applying Finland specific weights for each impact category to get the indicator value. The weights applied reflect the actualities of the different environmental impact categories as considered by a representative number of Finnish environmental experts in the beginning of 2000's [6].

Eco-Benchmark results presented in this paper are strictly limited to Finland. The impact rates used for normalisation and the weights given for different impact categories are not applicable to other countries or regions. The weighting of different environmental impacts depends on the state of the environment in the application region and changes with time. It should be also noted that all environmental impacts are not taken into account in the Eco-Benchmark indicator, because for some impacts there are no relevant methods available yet to make the assessment. Such impact categories include impacts of hazardous compounds and particulate emissions on human health, eco-toxic impacts, impacts on biodiversity, land-use, and impacts on the productivity of land.

3. RESULTS

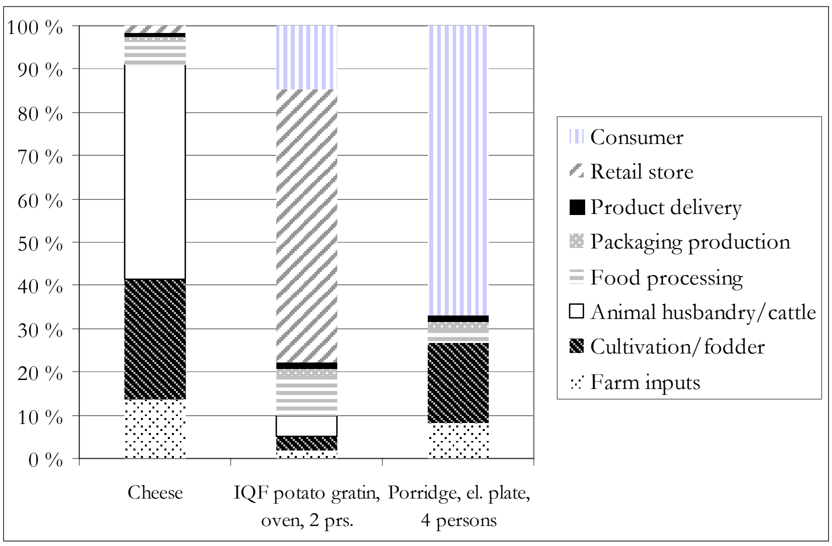


Figure 1. Contributions of life cycle phases for cheese, IQF potato gratin and porridge (oat meal) chains to global warming potential (GWP). Cooking is included.

The contribution of packaging to the global warming potential (GWP) of the entire food chain was 1 to 8% when cooking and other consumer related activities were not included. When cooking was included the contribution of packaging was around 1 to 2 % of the GWP of food product chains. GWP results of three case studies are shown in figure 1. However, in potato flour chain, packaging production had 8% share of GWP.

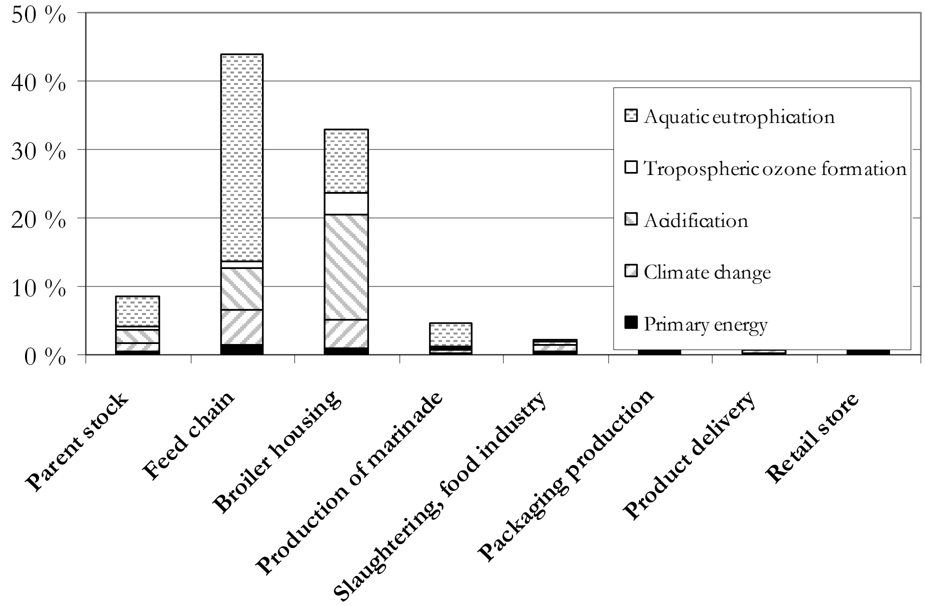


Figure 2. Contribution of the life cycle phases of the broiler fillet chain to the Finnish Eco-Benchmark value.

When the results of the different impact categories were combined to total environmental impacts using the Finnish Eco-Benchmark [6] method the contribution of the packaging production was set to less than 2%, as can be seen in figure 2, in which the distribution of the Eco-Benchmark value of broiler fillet chain by life cycle phases and impact categories is presented as an example.

Packaging production might have up to 15% contribution to the primary energy demand of total food product system. However, due to relatively large process specific emissions in other life cycle phases, e.g. nutrient and ammonia emissions from cultivation, the contribution of packaging to the other impact categories is much lower compared to the energy demand.

However, as a considerable exception to the results discussed above, in the chains of liquid products, such as beer, that mainly consist of water, packaging production have a higher relative environmental impact (excluding aquatic eutrophication) than in the chains of such products that contain little water. The contribution of the packaging to the total environmental impact (Eco-Benchmark value) of beer system was 21%. For the GWP beer packaging had around 30% share of climate change emissions. The packaging here included packaging (glass bottle and aluminium cans with corresponding market shares) production, production of primary materials, recovery of materials, delivery allocated to packaging and the content on mass basis.

Functionality of the product packaging affects the total environmental impacts of a food product chain. Minimising unnecessary product losses in the food logistics chains, retail and households, for instance, has a positive effect on the total environmental impact. On the other hand, a better functionality means changes in the impacts of the packaging. Packaging related impacts are often small in comparison to those of the actual food product. Thus, a rather small relative increase in the logistic yield, and a consequent reduction of the product related impacts achieved by changes in the packaging is enough to compensate for and overcome a much bigger relative increase in the packaging related impacts, and vice versa. The figure 3 demonstrates the impact balance between the change in the logistic yield and the packaging related impacts in food product chains.

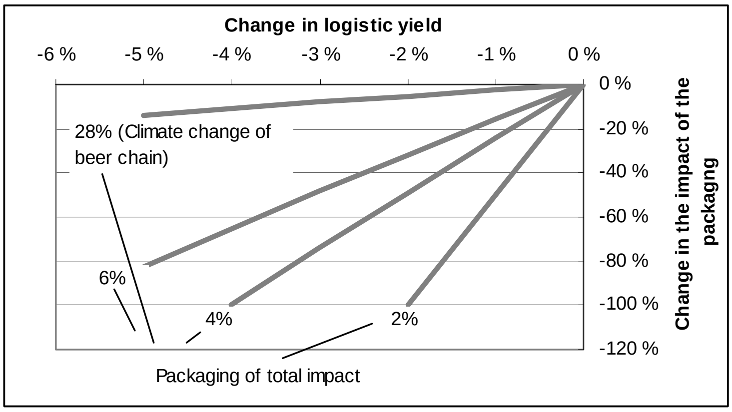


Figure 3. Equilibrium curves for the total environmental impact of a product chain when packaging makes 2%, 4%, 6%, and 28% of the total impact the reference system. Logistic yield is defined as the ratio of the delivered end-use amount to the produced amount.

For the reference system the change in the logistic yield = 0% and change of packaging related impact = 0%. On the curves total impact is equal to, left to the curves greater than, and right to the curves smaller than the reference value.

4. CONCLUSION

Based on the finding that the most of the environmental impacts of the food production chains were caused by the other activities than packaging, the present regulation and public interest that emphasise the packaging might not be targeted correctly from the environmental protection point of view. In fact, the role of the product packaging might be contrary to the common beliefs since packaging, when used effectively, might actually minimise environmental effects of the whole of the food product chains by avoiding unnecessary product losses in the food logistic chain, retail and households.

5. REFERENCES

1 J.-M. Katajajuuri, Y. Virtanen, P. Voutilainen and H.-R. Tuhkanen
Life cycle assessment results and related improvement potentials for oat and potato products as well as for cheese. Life Cycle Assessment in the Agri-food sector, Proceedings from the 4th International Conference, DIAS report no. 61, October 2004, p. 222-225. Bygholm, Denmark, October 6-8, 2003.

2 J.-M., Katajajuuri, H.-R. Tuhkanen and P. Voutilainen. Contribution of life cycle stages to the global warming potential of food products. Innovation by Life Cycle Management LCM 2005 International Conference, Proceedings, Volume I: p. 414-418. September 5-7, 2005, Barcelona, Spain.

3 J.-M. Katajajuuri, J. Grönroos and I. Sipilä. LCA of broiler production and related practical improvement options. SETAC Europe 17th Annual Meeting, 20-24 May 2007 Porto, Portugal. In print

4 Y. Virtanen, J.-M. Katajajuuri and K. Usva. Analysis of the Total Environmental Impact of Beer. 31st Congress of the European Brewery Convention, Venice, 6-10 May, 2007. Conference proceedings. In print.

5 Y. Virtanen, U. Ojaniemi, S. Poikkimäki and J-M. Katajajuuri. Life cycle assessment of potential environmental impacts of Finnish beverage packaging systems. 0.3 l-0.5 l glass bottles, aluminium cans and PET bottles for beer, cider and carbonated soft drinks. Summary Report. PTR report NO. 51a. 101 p.

6 A. Nissinen, J. Grönroos, E. Heiskanen, A. Honkanen, J.-M. Katajajuuri, S. Kurppa, T. Mäkinen, I. Mäenpää, J. Seppälä, P. Timonen, K. Usva, Y. Virtanen, and P. Voutilainen.
Developing benchmarks for consumer-oriented life cycle assesment-based environmental information on products, services and consumption patterns. Journal of Cleaner Production 15 (6): 538-549, 2007.

ACKNOWLEDGEMENTS

The results presented here are based on Finnish LCA studies *Foodchain (2000-2007)* and *Sustainability Tools of Malting Barley (2004-2007).* We wish to acknowledge all the companies and ministries involved for the financial support and active participation in the research.

How to include on-farm biodiversity in LCA on food?

Marie Trydeman Knudsen[a] and Niels Halberg[b]

[a]Faculty of Life Science, Department of Agricultural Science, University of Copenhagen, Hoejbakkegaard Allé 30, DK-2630 Taastrup, Denmark
[b]Faculty of Agricultural Sciences, Department of Agroecology and Environment, University of Aarhus, P.O. Box 50, DK-8830 Tjele, Denmark

1. INTRODUCTION

Life Cycle Assessments (LCA) of food and agriculture generally include potential effects on land use, global warming, eutrophication, ecotoxocity and acidification among other categories some of which again affect biodiversity. However, LCA most often does not include specific indicators of the product's or agricultural system's impact (negative or positive) on biodiversity. Using LCA methodology on agricultural products makes it highly relevant to assess the impacts of land use. Some LCA's include total land use per kg product, which is sometimes interpreted as "nature occupation". However, if this is the only impact category addressing land use related biodiversity, the LCA cannot distinguish between different forms of agricultural systems, which may differ in their biodiversity impact (e.g. organic versus conventional products) (Figure 1). Biologists as well as policy makers consider some agricultural land use, such as grazing semi-natural grasslands, as beneficial for biodiversity preservation.

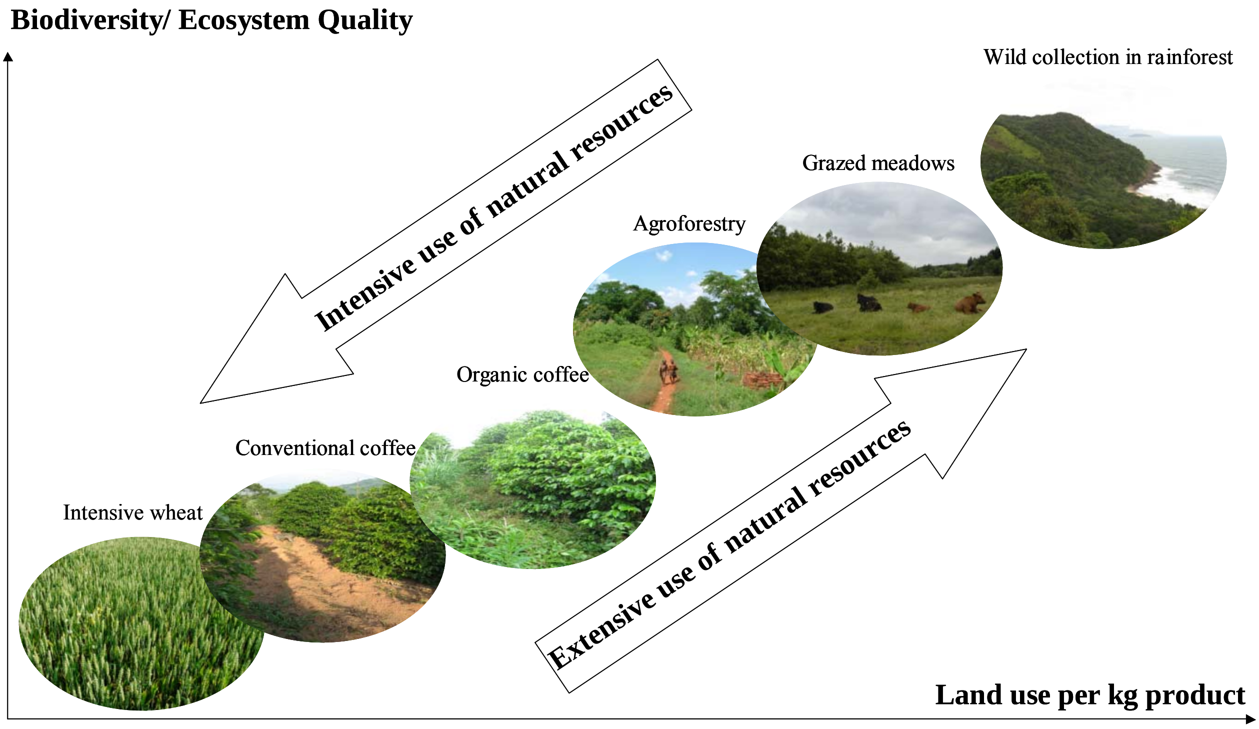


Figure 1. Illustration of the relation between land use per kg product and biodiversity/ecosystem value for different farming systems – indicating that the parameter land use per kg product can be difficult to interpret.

2. OBJECTIVE

To assess current approaches to include biodiversity aspects in LCA and search for an approach to include biodiversity aspects in LCA on food.

3. CURRENT APPROACHES TO INLUDE BIODIVERSITY IN LCA

Recently, attempts have been made to include effects on biodiversity and soil in a differentiated land use framework (Mila i Canals, 2007a) and a workshop on land use impacts in LCA has been held (Mila i Canals, 2007b). The challenge now is to search for simplified, comprehensive and operational indicators for biodiversity to include in LCA. Table 1 provides an overview of the different approaches suggested to include biodiversity aspects in LCA.

Table 1: Current approaches to include biodiversity in LCA

Indicators of biodiversity	Suggested by
Land use (ha year per kg product)	Current common LCA approach
Intactness, integrity, fragmentation, endemism, scarcity	Mila i Canals et al. (2007b)
Indicators based on ecosystem thermodynamics	Wagendorp et al. (2006)
The biotope method (four categories of biotopes)	Kyläkorpi et al. (2005)
Species richness indicator (SRI) & ecosystem rarity indicator (ERI)	Vogtländer et al. (2004)
The Hemeroby Concept (scale of use intensity, %)	Brentrup et al. (2002)
Several indicators especially on farmers uncultivated area	Schenck (2001)
Species richness (SR), Inherent ecosystem scarcity (ES), Ecosystem vulnerability (EV) – combined in Quality ($Q_{biodiversity}$)	Weidema & Lindeijer (2001)
Qualitative descriptions only	Mattsson et al. (2000)
Species-pool effect potentials (SPEP)	Köllner (2000)
Species diversity of vascular plants (S)	Lindeijer (2000)
Area, number of listed rare species, number of species, number of individuals	Cowell (1998)

4. THE SELECTION AND SUGGESTION OF ON-FARM INDICATORS

It is impossible to measure biodiversity directly in the farming systems for most LCA purposes and even measuring indicator species would be very time consuming and costly. Many current approaches to include biodiversity aspects in LCA include direct estimations of species diversity in the systems. A more simple and operational approach could be to identify the main causes for biodiversity impacts in the food product chain and develop simple proxy indicators for practical use in an LCA (for example so-called pressure indicators in the Driving-Force-Pressure-State-Response (DPSIR) approach (Smeets et al., 1999). At the farming systems level, questionnaires for farmers could thus be used in stead of measurements. In agroecosystems, such indicators for biodiversity could include the planned diversity (crop diversity, intercropping etc.), %uncultivated farm area, %area treated with pesticides, %weed in fields. An example of the practical use of the three last indicators is presented in Table 2 using data from Danish organic and conventional milk production. The indicators could later be aggregated into a single indicator describing e.g. “ecosystem quality” as suggested by Brentrup et al. (2002). However, challenges for such an approach are 1) how to aggregate different indicators consistently and without redundancy and 2) how to ‘translate’ such area based indicators into a product based assessment.

Table 2. Example of using several indicators on data from Danish

dairy farms (1994-1997).

	Organic	Conventional
Land use per kg milk (m^2 year)*	2.1	1.4
% small biotopes	4	4
% weeds in small grains	10	1
% unsprayed area	100	35

*LCA Food Database Source: Halberg et al. (!999)

5. CONCLUSION

Land use in food production systems can have both positive and negative impacts on biodiversity compared to leaving the land untouched by humans. Simple, operational indicators to account for the different impacts on biodiversity in food production systems could take the point of departure in the most important factors affecting biodiversity (easy obtainable pressure indicators) instead of estimating e.g. species diversity directly.

6. REFERENCES

Brentrup F, Küsters J, Lammel, J & Kuhlmann H (2002) Life Cycle Impact Sssessment of land use based on the Hemeroby Concept. International Journal of Life Cycle Assessment 7 (6): 339-348

Cowell S J (1998) Environmental life cycle assessment of agricultural systems: integration into decision-making. PhD dissertation, Guildford: Centre of Environmental Strategy, University of Surrey.

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

Köllner (2000) Species-pool effect potentials (SPEP) as a yardstick to evaluate land-use impacts on biodiversity. Journal of Cleaner Production 8: 293-311

Kyläkorpi L., Rydgren B., Ellegård A., Millander S. & Grusell E. (2005) The Biotope Method 2005 – a method to assess the impact od land use on biodiversity. Vattenfall.

Lindeijer E (2000) Biodiversity and life support impacts of land use in LCA. Journal of Cleaner Production 8: 313-319

Milà i Canals L, Bauer C, Depestele J, Dubreuil A, Knuchel R F, Gaillard G, Michelsen O, Müller-Wenk R & Rydgren B (2007a) Key Elements in a Framework for Land Use Impact Assessment Within LCA. International Journal of Life Cycle Assessment 12 (1): 5-15

Milà i Canals L, Clift R, Basson L, Hansen Y and Brandão M (2007b) Expert Workshop on Land Use Impacts in Life Cycle Assessment (LCA). International Journal of Life Cycle Assessment 11 (5): 363-368

Schenck R (2001) Land use and biodiversity indicators for Life Cycle Impact Assessment. International Journal of Life Cycle Assessment 6(2): 114-117

Smeets E, Wetering R, Bosch P, Büchele M, Gee D (1999) Environmental indicators: typology and overview. EEA Technical Report 25, 1-19

Vogtländer J G, Lindeijer E, Witte J-P M & Hendriks C (2004) Characterizing the change of land-use based on flora: application for EIA and LCA. Journal of Cleaner Production 12: 47-57

Wagendorp T, Gulinck H, Coppin P, Muys B (2006) Land use impact evaluation in life cycle assessment based on ecosystem thermodynamics. Energy 31: 112-125

Weidema B P & Lindeijer E (2001) Physical impacts of land use in product life cycle assessment. Final report of the EURENVIRON-LCAGAPS subproject on land use. Department of Manufacturing Engineering and Management, Technical University of Denmark, Lyngby, DK.Weidema B (2003): Market information in Life Cycle assessment. Environmental Project No. 863. Danish Ministry of the Environment. Environmental Protection Agency. Available at: http://www.mst.dk/homepage/default.asp?Sub=http://www.mst.dk/udgiv/publications/2003/87-7972-991-6/html/default_eng.htm

Environmental evaluation of excess pig slurry management

S. Lopez-Ridaura [a], H. van der Werf [a], J.M. Paillat [b]

[a] INRA, Agrocampus Rennes, UMR 1069, Sol Agronomie Spatialisation, F-35000 Rennes, France.
[b] CIRAD-INRA, Agrocampus Rennes, UMR 1069, Sol Agronomie Spatialisation, F-35000 Rennes, France.

Abstract
Slurry management plays a crucial role in the integration of crop and livestock production systems and the interaction between agriculture and the environment. This paper presents the comparison by Life Cycle Assessment of two scenarios of collective excess slurry management: The Transfer of slurry and its deep injection to crop land *vs* its Treatment in a collective biological treatment plant. The study is based on a case in Western France, where a group of farmers needs to dispose of more than 7000 m^3 of excess slurry. The overall environmental performance of the Transfer scenario is better than that of the Treatment scenario. The two scenarios are similar for Climate Change, whereas Eutrophication and Acidification are twice as large for Treatment relative to Transfer. Non-renewable Energy Use is 270 MJ m^{-3} for Treatment, whereas the Transfer scenario results in a net energy savings of 110 MJ m^{-3} due to the substitution of mineral fertilisers by slurry application to crops.

1. Introduction

Intensification of livestock production has generated new challenges related to the treatment and disposal of manure and slurry. Increased nutrient concentrations in underground and surface water threaten the ecological stability of intensive livestock production regions. Moreover, gaseous emissions (NH_3, N_2O and CH_4), resulting from animal buildings, manure and slurry storage and spreading on crop land, represent an important environmental burden associated with intensive livestock production.
Bretagne, in the West of France, concentrates 40% of the country's intensive livestock farming and it is one of the most polluted regions in France, specially with respect to nitrate in water, as organic and mineral Nitrogen (N) applied largely surpass crop needs. In some municipalities, and as part of the implementation of policies related to the Nitrate Directive, livestock farms exceeding a certain production of N as animal dejections must develop a plan for the disposal of the excess N in order to reduce its environmental impact by either treating the slurry or transferring it outside the region. A group of pig farmers have proposed a collective transfer and spreading plan where almost 7000 m^3 of slurry would be transported (over 40 km) and applied to crop land in substitution of mineral fertiliser.
The objective of this study is to evaluate the environmental performance of such a collective slurry transfer plan and compare it with a collective slurry treatment plant in order to contribute to a better understanding of the advantages and drawbacks of the two options.

2. Material and Methods

The environmental evaluation is carried out by Life Cycle Assessment as detailed in López-Ridaura et al; (2007). The functional unit of the study is 1 m^3 of slurry being either transferred or treated. The Transfer scenario includes: the on-farm slurry storage, its transport to the spreading area, its intermediate storage and its deep injection into crop land. Average distance between the pig farmers and the area receiving the slurry is 39.2 km and the transport is done with a semi-trailer truck equipped with a 25 m^3 cistern. Once in the spreading area, the slurry is temporarily stored in a flexible tank of PVC coated polyester (Figure 1-A). The Treatment scenario consist of 5 processes: on-farm slurry storage, its transport to the collective treatment plant, the treatment itself, the transport of compost (a by-product from the slurry treatment process) and the application of compost to crop land. Average distance between the pig farmers and the treatment plant is 12.1 km and the transport is done with a semi-trailer truck equipped with a 25 m^3 cistern (Figure 1-B).

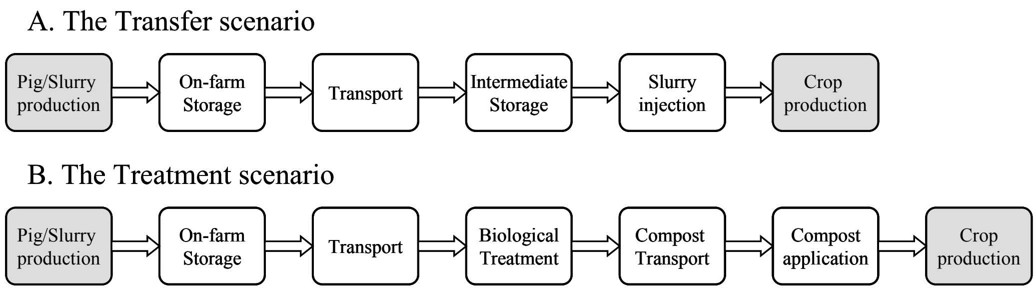


Figure 1. Schematic presentation of the processes involved in the two slurry management scenarios. Shaded boxes are not included in the study

The slurry treatment is of the aerobic or biological type (nitrification/denitrification), with previous separation of the solid and liquid fractions of the slurry with a centrifuge and the re-circulation of sludge. The solid fraction is composted for 9 weeks, involving the addition of 3% of straw and mechanical turning. Compost is then transported to a cereal production region at 200 km distance for its utilisation in substitution of fertilisers. The average abatement efficiency of the treatment is of 70% of the total N and 90% of the ammoniacal N (Loyon *et al.,* 2005).
For both scenarios, NH_3, N_2O and CH_4 emissions during storage and treatment are based on emission factors measured along the different stages of a biological treatment plant in Bretagne as reported in Loyon *et al.* (2005). Mass and nutrient balances were computed to calculate total gaseous losses at different stages of both slurry management scenarios and to know the characteristics of the resulting products.
Non-renewable energy use in the Transfer scenario includes the diesel used for the transport of slurry to the spreading area and for its injection. For the Treatment scenario it includes the diesel used for the transportation of slurry to the treatment plant, the electricity used for the treatment, and the diesel used for the transport and application of compost. The inventories for both scenarios include resource use and emissions associated with the production of the concrete and plastics (PVC, PET) needed for the storage and treatment of slurry as well as with the machinery needed for the injection of slurry and the spreading of compost.
As both raw slurry and compost are used in substitution of fertilisers, Mineral Fertiliser Equivalents (MFE) were calculated based on Morvan and Leterme. (2001). Total (direct, indirect and avoided) emissions and resource use are aggregated and expressed in terms of four impact categories: Eutrophication (in kg PO_4–eq.), Acidification (in kg SO_2–eq.), Climate Change (in kg CO_2 –eq.) and Non-Renewable Energy Use (in MJ of Lower Heating Value (LHV)-eq.) and were computed with SimaPro™.

3. Results

Figure 2 shows the comparison of the two scenarios for the four impacts normalised in relation to the values of the Treatment scenario. The overall environmental performance of the Transfer scenario is better than that of the Treatment scenario. The difference between the two scenarios is especially important for Eutrophication and Acidification as well as for Non-Renewable Energy Use.

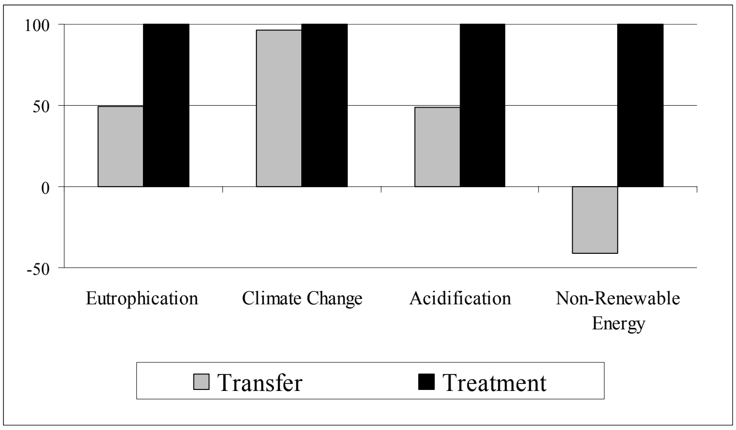


Figure 2. Environmental impacts of two slurry management scenarios expressed as a percentage of impacts for the Treatment scenario

Figure 3 shows the contribution of each phase along the Transfer and Treatment scenarios to the four impacts. NH_3 is the most important contributor to Eutrophication and Acidification for the two scenarios. In the Transfer scenario, on-farm storage is the main phase responsible for these impacts, while in the Treatment scenario it is the biological treatment phase, specifically during storage/homogenisation and the composting processes. In the Transfer scenario, the contribution of slurry injection to Eutrophication and Acidification is of similar magnitude than those avoided by the substitution of fertilisers, of which ca. 50% is due to NH_3 emissions during their application and the rest related to other substances emitted during the production and transportation of fertilisers (e.g. NO_2, P and SO_2). On the contrary, in the Treatment scenario, the substitution of fertilisers only compensates for ca. 30% of the impacts incurred during compost spreading due its low MFE.
For Climate Change, CH_4 is the major contributor for the two scenarios. As for the previous impacts, in the Transfer scenario, on-farm storage contributes most, while in the Treatment scenario the treatment phase, specifically during the storage/homogenisation and composting processes, is the major contributor. CO_2 is mainly emitted during transport of slurry and compost and N_2O is mainly emitted during the application of slurry, for the Transfer scenario, and compost, for the Treatment scenario.
The two scenarios strongly differ for Non-Renewable Energy Use,. In the Treatment scenario most energy is consumed as electricity during the treatment process, whereas the Transfer scenario represents an energy saving, as the energy needed for the production and transport of avoided chemical fertilisers is larger than that needed for the transport and injection of slurry. In fact, for a slurry with the characteristics of that used in this study (from finishing pigs, i.e. with high N and dry matter content), a transport distance of up to 87 km for its injection into crop land would allow maintaining an energy balance close to equilibrium.

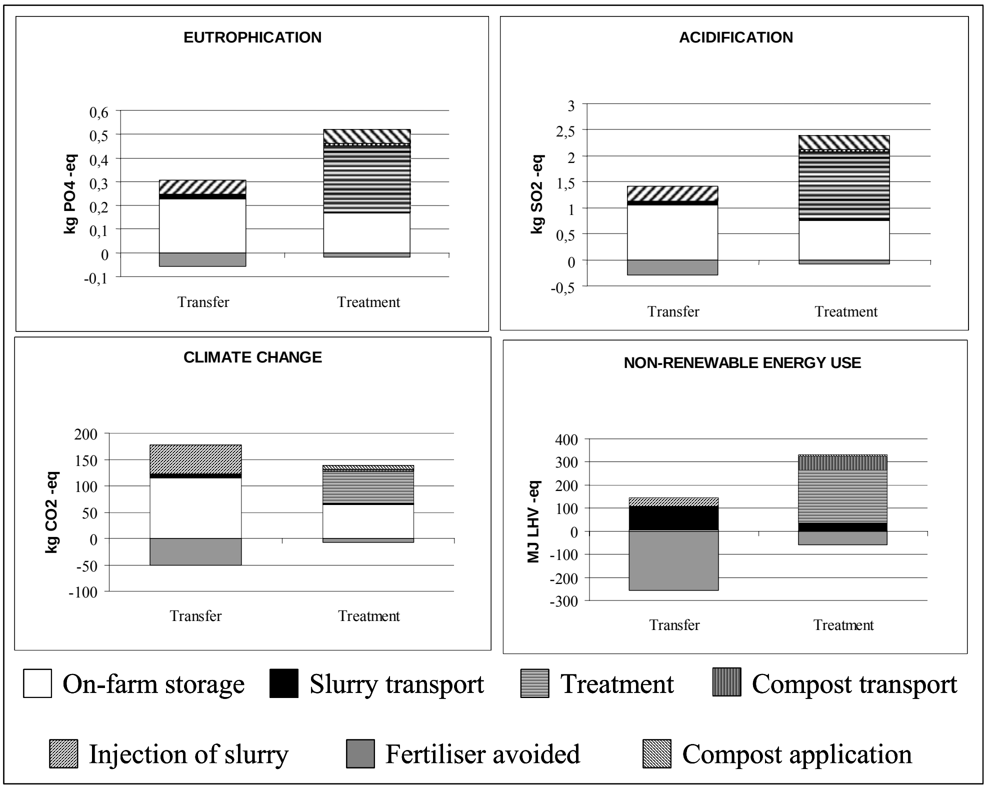


Figure 3. Contribution of the different stages of the Transfer and Treatment scenarios to four environmental impacts expressed per m^3 of slurry transferred or treated

4. Conclusions

For the two scenarios compared, the transfer of slurry and its injection into crop land in substitution of fertilisers poses less environmental threats than its biological treatment in a collective plant, as the former produces less gaseous emissions and consumes less non-renewable energy.

To refine the current analysis further research is needed including models predicting gaseous emissions and their uncertainty analysis as well as simulation models of storage tank dynamics because of the importance of storage time and slurry/atmosphere contact area in the gaseous emissions. Other alternatives for slurry management might also be worth exploring such as covering storage tanks, individual treatment plants, direct composting of slurry with the addition of different materials, the production of biogas for energy use and organic fertiliser for crops through methane digesters, different slurry and compost transport means, different application techniques of slurry to crops.

References

López-Ridaura, S., van der Werf, H., Paillat, J.M., le Bris, B. 2007. Environmental evaluation of transfer and treatment of excess pig slurry by Life Cycle Assessment. *Journal of Environmental Management.* (*Accepted*).

Loyon, L., Beline, F., Guiziou, F., Boursier, H., Peu, P. 2005. Bilan environnemental des procédés de traitement biologique des lisiers de porcs. Report ADEME-CEMAGREF, Rennes, France.

Morvan, T., Leterme, P. 2001. Vers une prévision opérationnelle des flux de N 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.

Life Cycle Assessment of Swiss integrated and organic farming systems

Thomas Nemecek, Olivier Huguenin-Elie, David Dubois & Gérard Gaillard

Agroscope Reckenholz-Tänikon Research Station ART, CH-8046 Zurich, Switzerland
thomas.nemecek@art.admin.ch

Abstract
The impacts on the environment of organic and integrated farming systems as well as of intensive and extensive management have been assessed in a comprehensive life cycle assessment study for Switzerland. The potential environmental impacts of organic farming (OF) were assessed more favourably compared to integrated production (IP). OF showed clear ecological advantages particularly for eco- and human toxicity, resource use and biodiversity. This positive assessment of OF only partly applies to nutrient losses and cannot be extended to single products in all cases. Per kg of organic product, higher impacts were often found for global warming potential, ozone formation, eutrophication and acidification compared to IP. No systematic differences compared to IP were found for soil quality for the same crop rotation and the same amount of organic fertilisers.

1. INTRODUCTION

Different methods of agricultural production respecting the environment have been developed during the last decades to solve the serious problems associated with intensive agriculture. These methods include integrated production (IP), organic farming (OF), extensive production and several environmental friendly techniques. The goal of this study is to make a comprehensive assessment of farming systems available for the Swiss arable crop and forage production. The following factors were investigated: farming system (conventional, integrated or organic), production intensity, procedures of fodder harvesting (cutting or grazing) and fodder conservation, form and quantity of fertiliser as well as the choice of arable crop and the production region (lowlands, hills or mountains).

2. OUTLINE OF THE STUDY

The study [1] evaluated the DOK and Burgrain farming system experiments (in Northern and Central Switzerland, respectively), as well as modelled arable crops and forage production systems. The DOK-trial covers the bio-dynamic, bio-organic and conventional/integrated farming systems, different fertiliser levels and organic and mineral fertilisation. In the trial Burgrain, the systems $IP_{intensive}$, $IP_{extensive}$ und organic were investigated. For the definition of the model arable crops and forage systems we used statistics, recommendations, documents from extension services, surveys, farm pilot networks and expert estimates. The 18 arable crops were differentiated according to the farming system, the intensity of crop protection and the production region (70 variants in total). For the forage production we analysed the following criteria: procedure of harvesting and conservation, duration of temporary meadow, production region, form of fertiliser, farming system, intensity of management and type of ecological compensation area (175 systems in total).
The life cycle assessments were carried out with the SALCA-methodology (Swiss Agricultural Life Cycle Assessment) of Agroscope Reckenholz-Tänikon (see [1]) by using the ecoinvent database [2]. The direct emissions of ammonia, nitrate, nitrous oxide, phosphorus and heavy metals, as well as the loss of soil, were calculated with models considering management and situation specific parameters. The system boundary was set to the farm gate for arable crop products and to the manger for the forage systems.

The analysis was performed in respect to three functions:

1. productive function (expressed as kg dry matter of harvested products, MJ net energy lactation (MJ NEL), g digestible protein or MJ gross energy)
2. land management function (hectare per year)
3. financial function (Swiss franc (CHF) gross profit).

3. KEY FACTORS IN FARMING SYSTEMS AND THEIR IMPACT

A correlation analysis showed close relationships between the different life cycle impact categories. These categories could be classified into three groups (Fig. 1): The *resource management* encompasses the energy demand, the global warming potential and the ozone formation. The *nutrient management* is represented by the eutrophication and the acidification. The aquatic and terrestrial ecotoxicity as well as the human toxicity can be summarised by *pollutant management*. The impacts on the *soil quality* and the *biodiversity* could be assessed by the newly developed SALCA-methods ([3] and [4]). The whole analysis could therefore be covered by these five environmental areas.

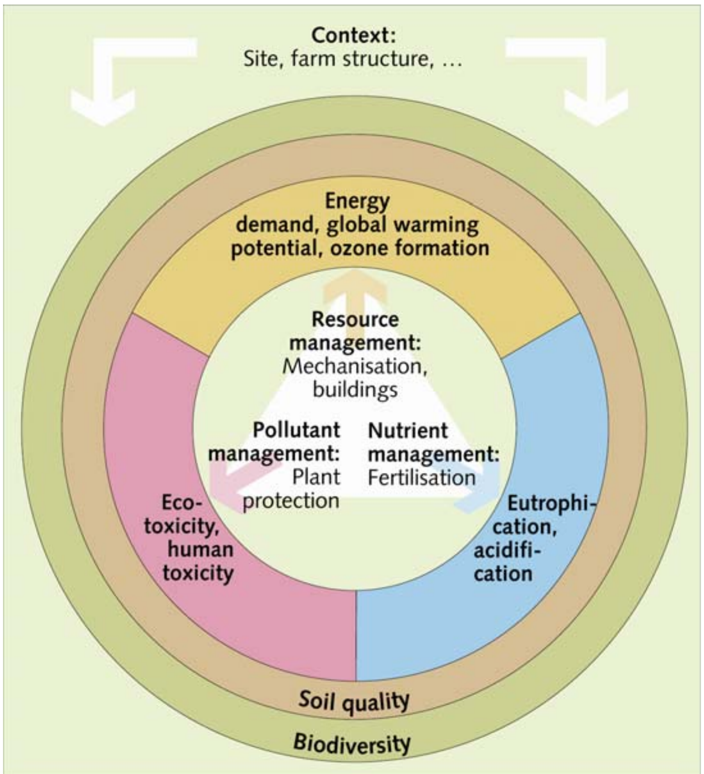


Fig. 1: Management triangle of farming systems (from [1]).

The potential environmental impacts of *organic farming* (OF) were on the whole favourably assessed compared to integrated production (Tab. 1). This is particularly the case for pollution management, resource management und biodiversity. However, the advantages of organic farming with respect to biodiversity cannot replace those of the ecological compensation areas. This positive assessment of organic farming only partly applies to the nutrient management and cannot be extended in all cases to single products. Per kg of organic product, higher values were often found for global warming potential, ozone formation, eutrophication and acidification compared to IP. No systematic differences to IP were found for soil quality for the same crop rotation and the same amount of organic fertilisers. The principal improvement needs for OF lie mainly in the increase of the yields – especially for the potatoes and the cereals – as well as the minimisation of nitrogen losses.

Conventional farming – analysed for wheat and rape seed – is clearly unfavourable especially for pollutant management by comparison with IP. For eutrophication and biodiversity the evaluation is also less favourable for conventional compared to integrated production.

The ban of fungicides, insecticides and growth regulators in cereals and rape seed (so called «*Extenso*» *production*) leads to an improvement in pollutant management and biodiversity. If considered per area, we found small advantages for resource and nutrient management. On the other hand, the product-related environmental impacts were often higher, due to the yield losses.

The *extensification of forage production* causes a significant reduction of environmental impacts per area unit. Extensive grassland is also environmentally more favourable per MJ NEL, but the differences depend on the considered impact category. A combination of plots managed at high and low intensity seems to be more environmentally favourable than the management of the whole grassland area at medium intensity.

Grassland management: Grazing results in a lower energy demand, but a higher global warming potential (higher emissions of nitrous oxide) than mowing. Eutrophication seems similar or higher for grazing than for mowing. The different types of conservation showed large differences for resource and pollution management: Silaging is environmentally more favourable than barn drying with ventilation. Sun drying of hay has lower environmental burdens than barn drying, although the latter can be improved by installing solar collectors.

Organic *fertilisers* use substantially less resources than mineral fertilisers and improve soil quality. On the other hand they have negative consequences for nutrient management, due to higher nutrient losses. A reduction of fertiliser input has positive effects with respect to almost all environmental impacts, when considered per area unit. The only exception is soil quality, where a reduction of organic fertiliser input can be harmful. Related to the product we found lower impacts mainly for nutrient management. Reducing fertiliser input can therefore have positive effects on the environment, but will probably reduce profitability.

Little differences in environmental impacts were found in respect to the *production region*. The impacts tended to be slightly lower at higher altitudes per area unit, with the exception of nutrient management. The contrary was true per kg of product. These small differences were caused by a lower production intensity and lower yields at higher altitudes.

4. CONCLUSIONS

The study revealed considerable improvement potentials for Swiss farming systems. Further improvement of the environmental performance of OF should focus on achieving higher yields of good quality – especially in potatoes and cereals, by using the limited inputs more efficiently – and on minimising nitrogen losses. For IP the main goal is to optimise input use in order to achieve a high eco-efficiency.

Reducing the management intensity leads to lower environmental impacts per hectare, but not always per product unit. Care must be taken that the whole farming system is considered in the optimisation process.

Tab. 1: Synthetic representation of the assessment of the impacts on the environmental of different factors (from [1]).

Factor:	Farming system					Production intensity				Production region		Form of fertiliser	Fertiliser quantity		Roughage harvest method	Roughage conservation method
Level:	organic			conven-tional		Plant prot.: extensive		Grassland management: extensive		Hills/mountains		Organic	Reduced		Pasture	Magnitude of differences
Reference:	vs integrated					vs intensive		vs intensive		vs lowlands		vs mineral	vs normal		vs mowing	
Functional unit:	ha	kg 1)	kg 2)	ha	kg	ha	kg	ha	NEL	ha	kg		ha	kg	ha or kg	
Resource management	++	++	+/-	0	0	+	-	+++	+	+	-	+++	+++	0	Energy+ GWP-	--/++
Nutrient management	++	0	+/-	-	0	0	-	+++	++	-	-	--	+++	++	0	0
Pollutant management	+++	+++	+++	---	---	++	++	+++	+	0	-	+	+	0	0	-/+
Biodiversity management	++			-		++		+++		+		0	++		Int- Ext+	0
Soil quality management	0			0		0				0		+++	-- 3)			

1) on farming system level
2) on crop level
3) for organic fertilisers

Int = intensive
Ext = extensive

Assessment classes:

---	Very unfavourable
--	Unfavourable
-	Unfavourable tendency
0	No relevant difference
	No assessment
+	Favourable tendecy
++	Favourable
+++	Very favourable
+/-	Favourable and unfavourable results

5. REFERENCES

1. Nemecek, T., Huguenin-Elie, O., Dubois, D., Gaillard, G., 2005. Ökobilanzierung von Anbausystemen im schweizerischen Acker- und Futterbau. Agroscope FAL Reckenholz, Zürich; Schriftenreihe der FAL 58, 155 p.
2. Frischknecht, R., Jungbluth, N., Althaus, H.-J., Doka, G., Hellweg, S., Hischier, R., Nemecek, T., Rebitzer, G., Spielmann, M., 2004a. Overview and Methodology - ecoinvent data v1.1. Swiss Centre for Life Cycle Inventories (ecoinvent), Dübendorf; ecoinvent report 1, 75 p.
3. Oberholzer, H.-R., Weisskopf, P., Gaillard, G., Weiss, F., Freiermuth, R., 2006. Methode zur Beurteilung der Wirkungen landwirtschaftlicher Bewirtschaftung auf die Bodenqualität in Ökobilanzen – SALCA-SQ. Agroscope FAL Reckenholz, Online at http://www.reckenholz.ch/doc/de/forsch/control/bilanz/salca-sq.pdf, 98 p.
4. Jeanneret, P., Baumgartner, D., Freiermuth, R., Gaillard, G., 2006. Méthode d'évaluation de l'impact des activités agricoles sur la biodiversité dans les bilans écologiques – SALCA-BD. Agroscope FAL Reckenholz, Online at http://www.reckenholz.ch/doc/fr/forsch/control/bilanz/salca-bd.pdf, 67 p.

LCI of must enrichment by reverse osmosis pilot plant

B. Notarnicola[a], G. Tassielli[a], P. Renzulli[b], E. Settanni[a]

[a]Department of Commodity Science, University of Bari, Italy.
[b] Renzulli Consulting Engineering, Bari, Italy.

Abstract
The enrichment of musts is one of the most spread practices in the wine making process. Today the method of concentration at high temperature is the most spread one. It requires great energy quantities and it can affect in a negative way the organoleptic characteristics of the product. On the contrary, reverse osmosis concentration methods are still poorly investigated in the wine making process even if they offer promising development prospects, mostly with regard to the semi permeable membranes to be used. Itest, a southern italian mechanical company, specialised in the building and set up of wineries, has developed a pilot plant for the study of the characteristics of the membranes to be used with native vines. In this paper, which is extracted from an ongoing research project financed by the Regione Puglia, the Life Cycle Inventory of the reverse osmosis plant for the must enrichment will be carried out.

1. INTRODUCTION

The enrichment of musts is one of the most spread practices in the wine making process. Its goal is to reach an optimal sugar level in musts, by increasing (within the limits of the Community rules) the content of reducing sugars and therefore the alcoholic grade of the wine so obtained. The enrichment made through the addition of sugary matters has been for a long time the most used practice for various reasons: simplicity of execution, economical benefit due to the Community contributions, increase in the total volume of the wine produced. Subtractive methods, on the other hand, have been used for producing concentrated musts or valuable wines, especially through the drying of grapes on the vine or after harvesting. The subtractive methods for the enrichment of musts suit the production needs of a sector that is experiencing a deep transformation. The subtractive methods allow the decrease of the volume of wine produced and avoid the introduction of components other than the must to be purified in order to increase its final alcoholic content. The EC Regulation 1493/99 [1] states that a maximum of two alcoholic degrees of enrichment and a reduction of the volume, not exceeding 20%, are allowed. Today the method of concentration at high temperature is the most spread one. Concentration is obtained through separation after evaporation in dedicated devices. It requires great energy quantities and it can affect in a negative way the organoleptic characteristics of the product, due to the loss of certain substances with a low boiling point, such as the primary aromas of grapes. Recently the method of low temperature concentration has become available on the market.
The reverse osmosis concentration methods are still poorly investigated in the wine making process, mostly with regard to the semi permeable membranes [2] to be used and various aspects such as energy, materials and environment of the plants, also considering the process variables. Reverse osmosis is based on the principle of separation of water through special semi permeable membranes that water molecules, but not solutes, can easily pass through. In this way, water is subtracted by must, thus not altering its physical state, operating at a temperature close to the room one. In the process, musts are subjected to high pressures and have to show a modest content of suspended solids, which could slow down or prevent the normal flow of the product. The reverse osmosis technique has notable advantages compared to other systems of enrichment: for instance, it seems to be more energy-efficient than concentration systems that use evaporation and also seems more able to preserve the organoleptic characteristics of the must to be treated. Moreover, little is known

about the environmental compatibility of the plants and the processes. Itest, a mechanical company of the South of Italy has developed a pilot plant for the study of the characteristics of the membranes, depending on the product to be treated.
The general goal of the ongoing research project is to acquire the knowledge necessary for assessing the environmental characteristics of the reverse osmosis plants for the enrichment of musts, which in turn is essential for the optimisation of the development of such plants. It has been agreed that the most suitable application methodology for this study is the Life Cycle Assessment of the product (LCA), standardized by the rules of the series ISO 14040 that can provide useful indications in the case of "Design for Environment" [3]. In this paper the LCI of reverse osmosis pilot plant has been carried out.

2. GOAL AND SCOPE DEFINITION

The "functional unit" of the study is the enrichment of 1000 L of must by 1 alcoholic degree. The examined system includes the extraction and the processing of raw materials, the production of the plant, transport and distribution, use, re-use and maintenance, recycling of the components and final disposal. The agricultural and winemaking phases have not been considered since the aim of this study focuses only on the process of must enrichment via reverse osmosis. The system to be studied is represented in Figure1

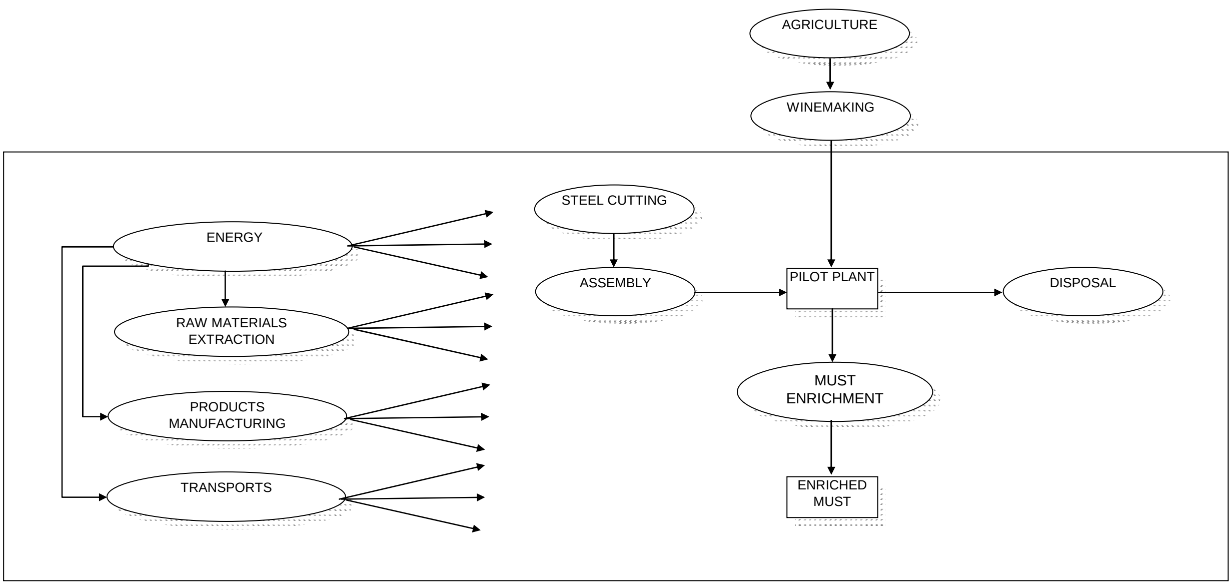


Figure 1. Diagram of the flow of the system to be considered.

3. LIFE CYCLE INVENTORY

The production cycle of the system is typical of metalworking industries. The primary material used is steel. The structural shaped steel that arrives at the production site is cut via band saw to obtain the desired shapes. Some of these components are drilled to prepare them for the next phase. The assembly phase involves soldering and joining via bolts components produced on site or purchased (membranes, monometers, pneumatic pressure switches etc.). The final system is made up of a structural part, an electrical system and a hydraulic one. Table 1 summarises the energy consumption, auxiliary material and products used in the production phase of the reverse osmosis plant. As can be noticed there are no particular impacts in this phase since the production is simple and only involves cutting and assembly operations. The largest effort on behalf of the firm is the research, design and development performed for this system. Table 2 shows the typical materials used subdivided for each main component. The mass of the system is approximately 440 kg; steel is the main material used with a mass of 358 kg, followed by the polyamide resins, that make up the filters, aluminium and copper.

Table 1: Energy consumption, auxiliary material and products used in the production phase of a standard plant.

Description	Units	Quantity
Cutting fluid	g	300
Welding rods (Aisi 304 steel)	g	500
Abrasive discs (phenOl resin/aluminium oxide: 85%/15%)	g	2000
Argon gas for welding	cubic meters	18
Electrical energy	kWh	58
Band saw blade (steel)	g	300
Drill bits HSS (steel)	g	100
Scrap left over Aisi 304 Steel	g	5000
Pickling paste	g	500

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

Description of components	Steel Aisi 304	Steel Aisi 316	Steel C40	Iron	NBR Nitrile Rubber	Silicon ic rubber	Nylon	Polyamide	PVC	PMMA	Brass	Copper	Aluminium alloy	Aluminium	Cast Iron	Poliurethane	Lubrication oil	Ceramic	Glycerine	Glass	EPDM for foods	
	Mass (g)	Mass (g)	Mass (g)	Mass (g)	Mass (g)	Mass (g)	Mass (g)	Mass (g)	Mass (g)	Mass (g)	Mass (g)	Mass (g)	Mass (g)	Mass (g)	Mass (g)	Mass (g)	Mass (g)	Mass (g)	Mass (g)	Mass (g)	Mass (g)	Total mass (g) per componen
Tubes	32640																					**32640**
Tube Joints	19249	1052			244	266							710									**21521**
Valves	6899	5089			5						37											**12030**
Accumulation pulsation damper		2807																			50	**2857**
Vessels with filters	180082					996	1572	42600														**225250**
Flow meters	503					1			8	293												**805**
Manometers		351							8										12	20		**390**
Pneumatic pressure switches		441							49													**490**
Eletricity controls	19566																					**19566**
Filters	6858				20	16																**6894**
Tanks	14508																					**14508**
Pumps	12041		33000	815		26	67		230		185	14230		16303	1955		300	441				**79592**
Steel plating	20588																					**20588**
Antivibration rubber					500																	**500**
Wheels	2156															764						**2920**
TOTALS (g)	**315090**	**9740**	**33000**	**815**	**769**	**1305**	**1639**	**42600**	**295**	**293**	**222**	**14230**	**710**	**16303**	**1955**	**764**	**300**	**441**	**12**	**20**	**50**	**440551**

The most relevant phase, with regards to this study, is the one during which the actual enrichment of the musts is performed via the machine representing the system. The separation of the water contained in the must is achieved via the polyamide filters that only allow the water particles to pass through them [4]. The filtering system consists of a sequence of vessels (each containing one of the above mentioned filters) connected in series through which the must is pumped in order to enrich it. For the filtering system to be effective the must has to be pumped into the filters at a pressure of approximately 65bars. The high pressure ensures that the (smaller) water particles pass across the filter and are thus collected separately from the enriched must (consisting of larger particles) that bypasses the filter and is pumped directly out of the vessels containing the filters and back into the machine for subsequent enrichment. Factors that influence the performance of the system are the

temperature of the surroundings, the temperature of the must, the initial and final alcohol grade, the kind of winemaking (red or white) and the kind of water used to wash the system before re-use. All these factors can lead to operational conditions that vary from a minimum of 70-80 L/h of water removed from the must to a maximum of 250-300L/h; the average is around 150-170L/h. The system consumes 5-6 kWh of electrical energy per hour and other materials such as lubricating oil.
The machine can work up to 48hrs non stop depending on the amount of suspended solid particles in the most/wine to be enriched. After this period the machine and the filters need to be washed with water and citric acid or potassium carbonate before being used again. Other less frequent maintenance tasks include the substitution of the mechanical filter, of the oil in the piston driven pump and of the filters every 1’000 hours of use of the machine. The lifetime of the machine is approximately 20’000 hours. From the data collected so far during the project, it is possible to indicate the energy and material consumption referred to the functional unit. It is necessary the plant to work 0.67 hours to realise the must enrichment of the functional unit.
The electric energy consumption is 0.0019 kWh and 4.02 kWh respectively for the production and use of the plant per functional unit; the materials consumption is 16.3 g and 28.5 g respectively for the production and use of the plant per functional unit. One can see that the actual must enrichment operation absorbs most of the direct energy and material consumption. The preproduction and production phases have a minor impact, which is typical for most of the studies on the life cycle of machines.

4. CONCLUSIONS

The state of the art of the analysis of the life cycle of systems for reverse osmosis for the enrichment of musts has led to the outline of an inventory of the direct material and energy consumption during the various phases of the examined cycle.
From these first results it is possible to show that the process of enrichment absorbs most of the direct energy and material consumption and that the preproduction and production phases have a minor impact which is typical for most of the studies on the life cycle of machines. A more detailed study of those extremely variable elements that contribute to the actual enrichment phase is therefore necessary together with an analysis of the final plant disposal/recycling data. Following this, it will be necessary to perform an assessment of the impacts related to the phases involved before the use of the system connected to the product and component supply and raw material extraction.

5. REFERENCES

1. European Commission: Council Regulation No 1493/1999 of 17 may 1999 on the common organisation of the market in wine. Official Journal of the European Communities, L 179/1, 14.7.1999.
2. A. Caetano, M.N. De Pinho, E. Drioli, H. Mantau: Membrane technology: Applications to Industrial Wastewater Traetment. Kluwer, The Netherlands, 1994.
3. B. Notarnicola, G. Tassielli, G. M. Nicoletti: LCA of wine production. In B. Mattsonn, U. Sonesson (editors): Environmentally-friendly food production, Cap. XVII. pg. 306-326. Woodhead-Publishing, Cambridge, England and CRC Press Boca Raton, USA, 2003.
4. P. Ribererau-Gayon, D. Dubourdieu, B. Donèche, A. Lonvaud : Handbook of Enology. Chichester, John Wiley & Sons, 2000.

Toxicity assessment of pesticides used in integrated pest management of orange crops in Spain

Nuno Martins[1], Ronnie Juraske[2], Neus Sanjuán[3]

[a]Escola Superior de Biotecnologia, Universidade Católica Portuguesa, Porto, Portugal
[b]Department of Chemical Engineering, ETSEQ, Universitat Rovira i Virgili, Tarragona, Spain
[c]ASPA Group, Department of Food Technology, Universitat Politècnica de València, València, Spain.

Abstract
The aim of this work was to assess the toxic impact of the pesticides usually applied in the integrated production of oranges in the *Comunidad Valenciana* (Spain), based on current LCA tools. The amount of the active ingredient applied according to the recommended practices, the fate of these ingredients into different environmental compartments, the human exposure routes and the inherent toxicity of them were taken into account in the calculations. The methodology used to asses the fate and exposure factors for human, and terrestrial and aquatic ecosystems was USES-LCA. Herbicides were the less toxic pesticide group for the three impact categories, while the pesticide group that generates the highest impact varied regarding the impact category. The results show that pesticide choice has a significant influence on the impact scores for the three impact categories. Consequently, the integrated production of oranges can be improved by a careful selection of the pesticides applied.

1. INTRODUCTION

Since the 80s, in Spain, national and regional governments have encouraged integrated farming thus aiming to develop more sustainable agricultural practices and to guarantee a better traceability of the products. In the last few years, the number of orange farms under the Integrated Production (IP) guarantee has increased in Spain, especially in the *Comunitat Valenciana* (CV). To be precise, in the citrus season 2005/06 there were 8.673,04 ha registered as IP.
Agricultural systems generate several impacts on the surrounding environment which are very important compared to industrial systems. The most relevant are: land use, water depletion and exposure to pesticides and subsequent toxic effects.
When calculating the impact that pesticides have on the humans and the environment there are some aspects that must be taken into account, such as (Milá i Canals et al., 2005): the quantity and application method of the pesticide, the transport of the pesticide through the different environmental compartments, the fate of the pesticide, the exposure to the pesticide and the effect on the population (humans and ecosystems) that is exposed to the pesticide.
This study was focused on the impact assessment stage of an LCA of the integrated orange production in the *Comunidad Valenciana* (Spain), specifically in the exposure to pesticides. For this aim the results of the inventory analysis were connected to the corresponding environmental impacts using USES-LCA as Life Cycle Impact Assessment (LCIA) method.

2. METHODOLOGY

A list of the pesticides allowed by the IP rules in the CV for the most common pests and the recommended dose for citrus trees was provided by specialists in this farming system.
The calculation of the impact score of the pesticides takes into account: the amount of active substance applied to the crop (Q), the transport factor from the crop to the environment (f), its fate

and exposure factor (F) and the effect (E) on the population. This can be expressed as (Guineé et al, 2001):

$$IS = \sum_{n}\sum_{x} Q_x \times f_{x,n\rightarrow j} \times E_{x,j} \qquad (1)$$

where IS, the impact to humans, aquatic ecosystems and terrestrial ecosystems and is represented by the human toxicity impact (HTI), aquatic toxicity impact (ATI) and terrestrial toxicity impact (TTI), respectively.

The different transport factors: wind drift, deposition on field soil and crop plants, volatilisation, surface run-off, leaching emissions to surrounding environment (water and soil), were calculated according to Antón (2004). To calculate the fraction of pesticide which leaches (f_l) from the top layer of the field soil and reaches the ground water, the GUS index (Vogue et al., 1994) was used in order to evaluate the potential movement of the pesticides. Fate and exposure factors were obtained from Huijbregts et al. (2005). The effects factors were calculated using the methodology proposed by Antón (2004), but for terrestrial ecosystems the no effect concentration (PNEC) was determined from the aquatic NEC using the equilibrium partition method (Huijbregts, 1999). All the impact scores were referred to the functional unit of the study, 1 kg oranges at the farm gate.

3. RESULTS AND DISCUSSION

The fraction that drifts off the field was estimated assuming that orange trees are developed crops and that the field where the pesticides are applied is entirely surrounded by ecosystems (aquatic and terrestrial). For this reason, the distance from the edge of the field is about 1 m, giving a result of 0.5 for the fraction of applied pesticide that drifts off the field. For herbicides, this amount was neglected as the substance is directly applied to soil (Milá i Canals, 2003). Consequently, herbicides do not neither reach the fruit and consequently the fraction that goes to the fruit is disregarded. For the surface runoff (f_r), a value of 0.0001 was used, because it was assumed that the slope of the field was irrelevant (Hauschild, 2000).

The impact score for each active substance in each category (humans, and aquatic and terrestrial ecosystems) using the USES-LCA is presented in Table 1. It can be observed that the active substances showing the highest impact on humans are Dicofol, Pyridaben and Propargite. Regarding the aquatic ecotoxicity, Malathion is the most toxic active substance followed by Clofentezine, Diazinon, Chlorpyrifos and Abamectin. Finally Diazinon, Propineb and Malathion are the pesticides with the highest impacts for terrestrial ecotoxicity.

Table 1- Impact scores for humans, aquatic and terrestrial ecosystems (USES-LCA)

Active substance	HTI (days·person^{-1}·FU^{-1})	ATI (days·FU^{-1})	TTI (days·kg$_{wwt}$·m^{-3}·FU^{-1})
Clofentezine	9,63E-05	4,03E-07	4,62E-07
Dicofol	1,71E-02	3,79E-09	5,39E-10
Fenazaquin	-	-	-
Fenbutatin-oxide	2,29E-11	6,32E-16	2,05E-17
Hexythiazox	3,81E-04	2,70E-11	2,21E-11
Propargite	6,47E-03	1,39E-08	3,91E-09
Tebufenpyrad	2,24E-03	1,13E-09	5,87E-11
Copper Oxychloride	-	-	-
Mancozeb	1,41E-06	1,88E-10	2,64E-11
Propineb	2,15E-04	9,35E-10	1,37E-06
Fluroxypyr	4,99E-10	7,37E-18	3,29E-15
Glufosinate-ammonium	3,44E-07	6,71E-17	2,50E-13

Glyphosate	7,21E-09	6,45E-15	5,52E-13
MCPA	-	1,10E-17	1,41E-13
Paraquat	4,25E-06	4,35E-15	3,71E-11
Abamectin	1,40E-03	2,11E-07	2,20E-08
Azadirachtin	-	-	-
Benfuracarb	1,93E-04	1,98E-10	5,75E-11
Buprofezin	8,46E-04	8,05E-11	2,51E-11
Carbosulfan	1,64E-03	3,07E-08	9,08E-09
Chlorpyrifos	2,46E-03	2,12E-07	2,96E-08
Diazinon	2,42E-04	3,26E-07	4,81E-06
Imidacloprid	5,55E-03	2,23E-09	3,60E-08
Malathion	3,85E-06	4,36E-07	9,64E-07
Pyridaben	7,03E-03	4,94E-08	7,12E-11
Pyriproxyfen	2,59E-04	1,09E-09	1,14E-11
Spinosyn A – Spinosad	-	-	-
Spinosyn D – Spinosad	-	-	-

The active substances generating the lowest impacts are Fenbutatin-oxide and Fluroxypyr for human toxicity, Fluroxypyr and MCPA for aquatic ecotoxicity and Fenbutatin-oxide and Fluroxypyr for terrestrial ecotoxicity.
With respect to the impact score results, human toxicity ranges from $2.29\cdot10^{-11}$ to $1.71\cdot10^{-2}$ days·person·FU^{-1}, in the case of aquatic ecotoxicity it ranges from $7.37\cdot10^{-18}$ to $4.36\cdot10^{-7}$ days·FU^{-1} and terrestrial ecotoxicity ranges from $2.05\cdot10^{-17}$ to $4.81\cdot10^{-6}$ days·$kg_{wwt}^{-1}\cdot m^{-3}\cdot FU^{-1}$.
These results give a general idea of the impact generated by pesticide application. Direct human exposure via orange consumption can not be modelled with USES-LCA, for this reason, the impact that pesticides have on humans via orange consumption was not considered in this study. Juraske et al. (2007) have developed a method that allows the assessment of this impact. Furthermore, this model does not take into account the specific conditions of the place under study. For example, the influence that temperature or rain may have on the fractions of pesticides that reach aquatic or terrestrial ecosystems. The wind speed is not taken into account like it is made by Antón (2004) and Antón et al. (2004). Another aspect that has not been considered is the possibility that a fraction of a pesticide which is applied in one farm, can travel to another farm.

4. CONCLUSION

From the results it can be concluded that herbicides is the pesticide group showing the lowest impacts on humans, aquatic ecosystems and terrestrial ecosystems. The most toxic pesticides are Insecticides for humans, Acaricides for aquatic ecosystems and Fungicides for terrestrial ecosystems.
These results give only a general idea of the impact generated by pesticide application due to the level of uncertainty of the method. Nevertheless, the present study shows that the choice of a pesticide for pest management has a significant influence on the impact scores for the three impact categories considered.

5. REFERENCES

Antón, A. (2004) *Utilización del Análisis del Ciclo de Vida en la Evaluación del Impacto Ambiental del Cultivo bajo Invernadero Mediterráneo* [PhD dissertation]. Polytechnic University of Catalonia: Barcelona

Antón, A., Castells, F., Montero, J. I., Huijbregts, M. (2004) Comparison of toxicological impacts of integrated and chemical pest management in Mediterranean greenhouses. *Chemosphere*, 54: 1225-1235

Guinée, J. B., Goree, M., Heijungs, R., Huppes, G., Kleijn, R., De Koning, A., Wegener Sleeswijk, A., Suh, S., Udo de Haes, H. A., De Bruijn, J. A., Van Duin, R., Huijbregts, M. A. J. (2001) *Life Cycle Assessment. Operational Guide to the ISO Standards*. Centre of Environmental Science, Leiden University (CML), The Netherlands,

Hauschild, M. (2000) Estimating pesticide emissions for LCA of agricultural products. pp 64-79. In: *Data for Life Cycle Assessment* (B.P. Weidema & M.J.G. Meeusen, editors). Agricultural Economics Research Institute (LEI), The Hauge, Wageningen.

Huijberts, M.A.J. (1999) *Ecotoxicological effect factors for the terrestrial environment in the frame of LCA*. Interfaculty Department of Environmental Science, Faculty of Environmental Sciences, University of Amsterdam.

Huijbregts, M.A.J., Struijs, J., Goedkoop, M., Heijungs, R., Jan Hendricks, A., Van de Meent, D. (2005) Human population intake fractions and environmental fate factors of toxic pollutants in LCIA. *Chemosphere* 61:1495-1504

Juraske, R., Antón, A., Castells, F., Huijbregts, M.A.J. (2007) Human Intake fractions of pesticides via greenhouse tomato consumption: comparing model estimates with measurements for Captan. *Chemosphere* 67:1102-1107

Milà i Canals, L., Antón, A., Sanjuán, N. (2005) Aspectos metodológicos del ACV agricola. pp 79-88. In: *Análisis de Ciclo de Vida: Aspectos Metodológicos y Casos Práticos* (G. Clemente, N. Sanjuán & J.L. Vivancos, editors). Editorial Universidad Politécnica de Valencia, Valencia

Vogue, P.A., Kerle, E.A., Jenkins, J.J. (1994) OSU Extension Pesticides Properties Database. Available: http://npic.orst.edu/ppdmove.htm [data visited: 23/1/06]

Introduction of the Food Study Group, the Institute of Life Cycle Assessment, Japan

Toshisuke OZAWA[a] and Atsushi INABA[a,b]

[a]Research Center for Life Cycle Assessment
National Institute of Advanced Industrial Science and Technology (AIST)
16-1 Onogawa, Tsukuba, Ibaraki 305-8569 JAPAN
[b]Research into Artifacts, Center for Engineering (RACE), The University of Tokyo
Kashiwanoha 5-1-5, Kashiwa, Chiba 277-8568, JAPAN

Abstract
The Food Study Group (FSG) has carried out research and surveys in the Institute of Life Cycle Assessment Japan to identify the possible directions to sustainable food consumption and production. FSG has mainly two tasks: (1) life cycle inventory analysis on food products and meals; and (2) determine the food values to develop a sustainability indicator for agri-food consumption and production. For the former task, FSG evaluated life cycle CO_2 emission for various agri-food products including grains, meat, and vegetables, and for five different meals that consist of those products as ingredients. The result suggested that the CO_2 emissions per kg-product for high protein products (meat, dairy, and fish) and carbohydrate products (rice and bread) tend to be high. With regard to cooking, boiling and steaming tend to emit more CO_2 than stir-frying and deep-frying due to their longer cooking time.

1. INTRODUCTION

In order to turn the on-going agri-food consumption and production pattern into a more sustainable mode, quantification of environmental load on concerning products throughout their entire lifecycle is a prerequisite. Moreover, it is necessary to seek acceptable measures in consumption and production patterns that could lead to sustainability. In order to evaluate the effectiveness of the adopted sustainability measures, a sustainable indicator is necessary. In this paper, the concept of eco-efficiency (WBCSD, 1992) was adopted for the development of a sustainability indicator. Eco-efficiency is evaluated by comparing a concerning product's service value with its environmental loads. The measures that lead to sustainable food system should either: (1) minimise the environmental burden caused by food; or (2) maximise the value of food for consumers, or both. Life cycle assessment (LCA) is used to quantify the environmental burden from food items and meals.

The Food Study Group (FSG), a voluntary-based study group, is sponsored by the Institute of Life Cycle Assessment, Japan. The director of FSG is Dr. Atsushi Inaba of University of Tokyo, and membership includes 40 from various organizations, including academia, national research institutes, consulting companies, and private enterprises. FSG had mainly two tasks for 2005-2006: (1) life cycle inventory analysis on food products and meals; and (2) determine the food values to develop a sustainability indicator for agri-food consumption and production. Through these tasks, FSG aims to investigate each production process of agri-food materials to find new research themes for the solutions to the current environmental issues. FSG set forth five meal menus, accumulated inventory data and calculated life cycle CO_2 emission (LC-CO_2) for major agri-food products and for five different meals that consist of those products as ingredients. The purpose of this paper is to introduce FSG's output from our life cycle inventory analysis. Details on the outcome of Task 2 are described elsewhere (Ozawa *et al.*, 2007).

2. SETTING MODEL MENUS

The model menu for the calculation of LC-CO_2 was set based on the consideration of the following: (1) include staple food, main dish, side dish, soup, and high frequency in appearance in the average household's tables; and (2) with a variety in styles in cooking, including home cooking, processed food and restaurant. In this way, assessment of the difference in LC-CO_2 of the same dish in the same menu is available by altering from home cooked meal to retort or processed meal from a convenience store. The menus set for this study are shown in Table 1.

Table 1 The model menus set for this study

Breakfast	Lunch + Sweets	Dinner 1 (Japanese)	Dinner 2 (Western)	Dinner 3 (Chinese)
Toast	Ramen noodle	Rice	Rice	Rice
Fried egg	Soup	*Miso* soup	Corn potage soup	*Zya zuai row su tan*
Salad	Fruit	Grilled fish	Hamburg stake	Fried chicken
Yogurt	Tea	*Chawan mushi*	Bean saute and Carrot grasset	Stirred vegetables with thick sauce
Coffee	Blancmange	Pickles	Potato salad	Desert
		Fruits	Fruit	Beer
		Beer		

The amount of each ingredient was determined based on the Health, Labour and Welfare Ministry's "Japanese Food Guide Spinning Top." The Guide illustrates the applicable amounts of food items per serving for Japanese adults to stay healthy. We made sure that the amounts of nutrients taken through the model menu are sufficient to the demand of women 30-49 in age with the activity level of "average" (Class II). Then, the weights of each ingredient were calculated into gram-use per four persons, assuming that the average household has four family members.

3. RESULTS

3.1. LCI of Food Items via Process LCA

The output of FSG in 2005-2006 includes calculation of LC-CO_2 for various agri-food products, such as rice, wheat, soybeans, crude sugar, tomato, dried noodle, vegetable oil, refined sugar, cooked rice and meat, and for five different meals that consist of those products as ingredients. A hybrid approach of process LCA and I-O analysis was taken. The survey on imported goods included energy consumption at production stage in the country of origin and ocean transportations.

Table 2 shows the results of LCI of agricultural products via process LCA. In the case of imported wheat from the United States arriving to Japan, the LC-CO_2 is estimated to be 383.8 g-CO_2/kg-wheat. The LC-CO_2 of flour based on the imported wheat is estimated to be 520.1 g-CO_2/kg-flour. The LC-CO_2 for other imported items are: soybeans 413 g-CO_2/kg-product; and unrefined sugar 232 g-CO_2/kg-product. The domestic products are: brown rice 333, cabbage 39, and tomato (758 and 179) without transportation (in g-CO_2/kg-product). Notice that the green house grown tomato costs over four times as much CO_2 as that of tarpped.

Table 3 shows CO_2 emissions of various processed food products. For example, the CO_2 emission of bread is 1,013 g-CO_2/kg-dry (251.2 g-CO_2/loaf). In Japan, a slice of toast is served at a meal and is 1/6 of a loaf. Therefore, the CO_2 emission from one serve of toast is 42 g-CO_2. It is clear from the results in Table 3 that the CO_2 emissions for one serve of toast, wet noodle (49.4 g-CO_2/kg-product) and dried noodle (68.2 g-CO_2/kg-product) are in the comparable range.

Table 2 CO_2 emissions of agricultural products (unit: g-CO_2/kg-product)

	Wheat	Soybeans	Imported unrefined sugar	Rice (brown rice)	Cabbage	Tomato	Tomato
	U.S.A.	U.S.A.	Thailand	Domestic	Domestic	Green house	Tarpped

Agricultural process abroad	206	197	146				
Inland transportation	84	134	38				
Marine transportation	94	82	48				
Agricultural process in Japan				333	39	758	179
Transportation				59		152	152
Total	384	413	232	392	39	910	331

However, the CO_2 emission from imported wheat is predominant in noodle production, whereas both wheat and baking are dominant in bread production, indicating that baking process is more energy intensive than noodle drying or frying. The CO_2 emissions of soybean oil, refined sugar and steamed rice are in the comparable range from 580 to 850 g-CO_2/kg-product. Further, calculation on beef cattle production revealed its LC-CO_2 to be 10.9 kg-CO_2/kg-beef (data not shown). The results of the inventory analysis suggested that the LC-CO_2 were higher in protein-rich products followed by carbohydrate-rich products.

Table 3 CO_2 emissions of various processed food products (Unit: g-CO_2/kg-product)

Bread baking
FU: A loaf of bread

	Emission (g-CO_2)	Ratio (%)
Imported Wheat	116	46.2
Flour Milling	11.7	4.7
Other Ingredients	37	14.7
Bread Baking	86.5	34.4
Total	251.2	100

Wet noodles
FU: One serve (120g)

	Emission (g-CO_2)	Ratio (%)
Imported Wheat	41.8	84.6
Flour Milling	4.1	8.3
Other Ingredients	1	2
Noodle Making	2.5	5.1
Total	49.4	100

Dried noodles
FU: One serve (100g)

	Emission (g-CO_2)	Ratio (%)
Imported Wheat	46	67.4
Flour Milling	4.5	6.6
Other Ingredients	3.2	4.7
Noodle Making	14.5	21.3
Total	68.2	100

Soybean oil production

FU: 1 kg-product

	Emission (g-CO_2)	Ratio (%)
Imported Soybeans	432.7	62.4
Soybean Meal Production	73.4	10.6
Degumming	103.1	14.9
Refining Soybean oil	84	12.1
Total	693.2	100

Refined sugar production

FU: 1 kg-product

	Emission (g-CO_2)	Ratio (%)
Imported Unrefined Sugar	239	41.1
Refining Sugar	342	58.9
Total	581	100

Rice production and cooking

FU: 1 kg-product

	Emission (g-CO_2)	Ratio (%)
Domestic Production	370	43.8
Polishing and Packaging	12	1.4
Transportation	54	6.4
Rice Cooking	398	47.1
Disposing	11	1.3
Total	845	100

3.2. LCI on Meals via Hybrid LCA

CO_2 emissions from house-cooked meals were calculated by combining the CO_2 emission of the ingredients and that of direct energy consumption through cooking. The CO_2 intensity data for ingredients are taken from 3EID based on I-O table in 1995. For some food items, we chose to use the inventory data that were obtained from our process LCA. The LC-CO_2 on a model meal is shown in Table 4. The columns from left to right include ingredients and weights, CO_2 intensities of ingredients and energy consumption via cooking, calculated CO_2 emission via process LCA for some food items, respectively. Table 5 shows the summary of LCI results of the three model meals. Among the model meals, the LC-CO_2 including ingredients and cooking with I-O data and process LCA for breakfast was the smallest (1,172 g-CO_2), and lunch closed to 2,000g-CO_2, dinner (Japanese and Chinese) 3,000 g-CO_2, and the highest emission was found in Western dinner at 5,900 g-CO_2.

Table 4 LCI on model meals (Dinner 2-Western)

Dish	Ingredient	gram-use for four persons	Raw Code	Item in I-O	Unit Price Yen/Unit	Unit	t-CO2/Unit	LCI for CO2(4people) g-CO2	CO2 via Cooking g-CO2
Rice	Rice	400	1114-011-002	Rice domestic	467.5	t	0.791	316.4	79.6
									(HI Elect.)
Hamburg stake	Ground meat (beef)	300	1111-011-101	Beef, female	1,433,000	t	3.050	915.1	
	Bread crumb	20	1115-021-001	Bread	495,468	flour, t	1.103	14.6	
	Milk	45	1112-041-001	Milk	181,700	kl	0.487	21.9	
	Onion	80	0113-010-206	Onion	83,939	t	0.183	14.6	
	Vegetable oil	12	1117-041-101	Soybean oil	103,078	t	0.369	4.4	(Gas)
	Salt	3.6		Salt			-		
	Nutmeg	0.6		Nutmeg			-		
	Pepper			Pepper			-		(Gas)
	Egg	40	0121-021-001	Chicken egg	160,461	t	0.430	17.2	(fry)
	Butter	12	1112-042-301	Butter	950,080	t	2.545	30.5	
	Vegetable oil	12	1117-041-101	Soybean oil	103,078	t	0.369	4.4	
(sauce)	Butter	18	1112-042-301	Butter	950,080	t	2.545	45.8	
	Flour	14	1114-021-105	Flour	144,233	t	0.324	4.5	
	Tomato puree	30	1117-051-202	Tomato puree	288,649	t	0.887	26.6	
	Bouillon	21.2	1119-021-003	Soup	633,288	t	1.577	33.4	
	Salt	1		Salt			-		(Gas, boil)
	Pepper	a little		Pepper			-		
	Bay laurel			Bay laurel			-		
	Parsley			Parsley			-		
								Sub Total	92.2
Carrot grassed	Kidney beans	280	1113-010-112	Green bean	511,189	t	1.114	312.0	49.0
	Butter	20	1112-042-301	Butter	950,080	t	2.545	50.9	(Gas, fry)
	Carrot	200	0113-010-303	Carrot	129,592	t	0.282	56.5	53.8
	Butter	20	1112-042-301	Butter	950,080	t	2.545	50.9	(Gas, fry)
	Sugar	30	1117-011-201	Imported refined sugar	142,899	t	0.850	25.5	
	Salt	1		Salt			-		
Potato salad	Potato	300	0112-012-001	Potato	47,960	t	0.079	23.7	
	Lettuce	10	0113-010-211	Lettuce (ground)	97,510	t	0.213	2.1	(Gas, fry)
	Petit tomato	25	0113-010-107	Tomato (ground)	241,612	t	0.527	13.2	
	Cucumber	100	0113-010-103	Cucumber (ground)	209,372	t	0.456	45.6	
	Carrot	35	0113-010-303	Carrot (ground)	129,592	t	0.282	9.9	
	Mayonnaise	40	1117-051-401	mayonnaise	492,634	t	1.514	60.6	(Gas, fry)
	Salt	1		Salt			-		
	Pepper	a little		Pepper			-		
								Sub Total	136.7
Corn potage soup	Canned sweet corn	120	1116-011-203	canned sweet corn	249,120	t	0.877	105.2	
	White butter roux	16	1112-042-301	Butter	950,080	t	2.545	40.7	
	Flour	24	1114-021-105	Flour	144,233	t	0.324	7.8	(Gas, fry, boil)
	Bouillon	21.2	1119-021-003	Soup	633,288	t	1.577	33.4	
	Milk	240	1112-041-001	Milk	181,700	kl	0.487	116.8	
	Salt	2.4		Salt			-		
	Pepper			Pepper			-		
	Crouton	8	1115-021-001	Bread	495,468	flour, t	1.103	8.8	
	Vegetable oil	0.56	1117-041-101	Soybean oil	103,078	t	0.369	0.2	(Gas, fry)
	Parsley			Parsley			-		
								Sub Total	91.0
Fruit	Mixed fruit can	520							
	Apple	130	0114-012-101	Apple	149.201	t	0.1951	25.4	
	Grapefruit	208	0114-011-103	Orange	175.154	t	0.22906	47.6	
	Strawberry	182	0114-019-108	Strawberry (ground)	597,032	t	1.301	236.8	
Beer	Beer	1400		Beer	1.436	Mill. Yen	1.436	?	
								2,407.0	422.7
								Grand Total LC-CO2	2,829.7

Dish	Raw Data Unit	LCI for CO2(4people) g-CO2	
Rice	430.6	172.2	430.6×0.4 =
	(transportation from production to consumer)		Tahara, 2005
Hamburg stake	10,600	3,180	Ogino, 2006
	222.3 gCO2/loaf	11.1	Flour: Okumura, Sato, Ozawa, 2005 Bread: Okamoto, 2006 20g/400g 4 persons/loaf
	Hamburger in I-O 523,761	t	502.2g×0.9 = 1.308 623.7
(sauce)			
Carrot grassed			
Potato salad	138.67 g-CO2/kg	3.5	Shina, Okadome, Mishima, Saito, 2005
Corn potage soup	222.3 gCO2/loaf	4.4	Flour: Okumura, Sato, Ozawa, 2005 Bread: Okamoto, 2006 8g/400g
Fruit	g-CO2/kg		
Beer	364.9	510.9	231g-CO2/large bottle: Atsumi, 2005
	New LCI	5,432.4	
	Grand Total LC-CO2	5,855.1	

Table 5 Summary of LCI results on meals (unit: g-CO_2/meal)

	Ingredients only		Energy Consumption via cooking	Ingredients + Cooking	
	I-O Data only	I-O Data + Process LCA		I-O Data only	I-O Data + Process LCA
Breakfast	1,160	1,056	116	1,276	1,172
Lunch	1,706	1,167	773	2,479	1,940
Dinner 1 (Japanese)	2,060	2,571	637	2,754	3,084
Dinner 2 (Western)	2,407	5,432	423	2,830	5,855
Dinner 3 (Chinese)	2,591	2,664	295	2,885	2,958

4. CONCLUSIONS

1) Agricultural food products: high LC-CO_2 for proteins (livestock and marine products) and for carbohydrates (rice, bread, noodles),
2) Energy consumption in cooking: higher CO_2 for boiling, steaming, and simmering, than flying and deep-flying (Japanese: 600g-CO_2, Western: 400g-CO_2, Chinese: 300g-CO_2),
3) Meals: Breakfast - 1,000g-CO_2, Lunch - 2,000g-CO_2, Dinner (Japanese; Chinese) - 3,000g-CO_2, (Western) - 6,000g-CO_2,
4) LC-CO_2 via I-O Table Analysis: though there are some difficulties, the data are useful to acquire a holistic view of the agri-food production and consumption.

5. REFERENCES

1) World Business Council for Sustainable Development (WBCSD), (1992), "Changing Course," the report to Rio Summit.
2) Toshisuke OZAWA, Kiyotaka TAHARA and Atsushi INABA, (2007), *"Development of a Sustainability Indicator for Agro-Food Consumption and Production"* in the same proceedings.
3) Japanese Food Guide Spinning Top" (2005), Ministry of Health, Labour and Welfare, and Ministry of Agriculture, Forestry and Fisheries.
4) Embodied Energy and Emission Intensity Data for Japan Using Input-Output Tables (3EID): Inventory Data for LCA," Dr. Keisuke Nansai Research Center for Material Cycles and Waste Management, National Institute for Environmental Studies.
5) Ajinomoto version of LC-CO_2 coefficient database based on the Embodied Energy and Emission Intensity Data for Japan Using Input-Output Tables (3EID) and Input-Output Table Analysis (1995), Ajinomono Group Environmental Report, 2006 (URL: http://www.ajinomoto.co.jp/company/kankyo/report/pdf/p069-080.pdf).

LCA of “Spanish-style” green table olives

G. M. Nicoletti, G. M. Cappelletti, C. Russo[a]

[a]**Dipartimento di Scienze Economico-Aziendali, Giuridiche, Merceologiche e Geografiche**
Università degli studi di Foggia – Via Caggese 1 – 71100 Foggia – Italy

Abstract
The LCA methodology has been applied to highlight the environmental *hot spots* of the production steps of green table olives produced by the most common method used in the world *Spanish-style*. The agricultural practices have also been taken into account.
The analysis of the input and output material and energy flows has enabled us to propose a hypothesis for reducing the impacts on the environment.

INTRODUCTION

In the Mediterranean area the cultivation and processing of table olives plays a very important part in the region’s agro-industrial system.
The production of table olives has increased world-wide (from 1,343,000 tons in 2000/01 to 1,785,500 tons in 2005/06) [1].
The countries with the largest production of table olives are Spain (437,000 tons), Turkey (280,000 tons), Egypt (200,000 tons) and Greece (123,000 tons). In 2005/06 Italy produced 70,000 tons, while the amount of table olives consumed was 139,000 tons.

1. GOAL AND SCOPE DEFINITION

The subject of our study was the life-cycle of green *Spanish-style* table olives. Both the agricultural phase and the industrial phase were examined, with input and output flows represented by a functional unit equivalent to 100 kg. of harvested olives.
The agricultural phase includes all the agronomic practices connected with managing and irrigating an olive-grove, where the planting-space for each tree is 5 m * 6 m. The plantation is presumed to be in the South of Italy, and so the choice of fertilizers and plant-protection products was made with reference to the climatic conditions of this area [2].
As for the industrial phase, the most widely-used *Spanish-style* method was the one selected for examination. This method involves a process of deamarization, where the olives are kept in lye (1.5-3 % NaOH) for 8 - 12 hours and then rinsed several times with water. This is followed by a 2 - 3 month period of fermentation, when the olives are kept in brine (NaC1 6 - 8 %). The processes of packaging in brine and pasteurization were excluded from the study.

2. INVENTORY ANALYSIS

The data concerning the agricultural phase was obtained from local producers. Air, water and soil emissions resulting from the use of fertilizers and plant-protection products were assessed using models present in the literature [3-4].
As for the industrial phase, the data concerning energy use and raw materials was calculated as the average of all the data collected from local producers. For the analysis of waste liquids, reference was made to information available in the literature (table1) [5].
Our data was processed with the aid of GaBi4 software [6] using its databases (PE, Buwal250, Ecoinvent).

Table 1: Pollution charge in the lyes and washing waters of Spanish-style green table olives

Type of waste-water	pH	BOD_5	COD	NH_3	B	P	Cu	Zn	Ni	Hg	Cl^-	Na
		(g O_2/l)		(mg/l)								(g/l)
Lye	12.7	4.8	24.0	1.3	0.14	18.3	0.23	0.48	0.12	0.08	2.9	96.5
1st washing	9.1	3.1	10.0	16.8	0.63	28.4	0.16	0.36	0.09	0.05	1.9	15.7
2nd washing	7.6	1.6	7.6	1.3	0.40	16.3	0.10	1.64	0.03	0.05	1.1	10.5

Reference 5

When analysing samples of resources as well as emissions resulting from the cultivation and processing of the olives, the agricultural and industrial phases were examined separately; this enabled us to evaluate the amount of pollution for each phase.
Figure 1 shows the input quantities and operations used for the production and successive processing of 100 kg of harvested olives.

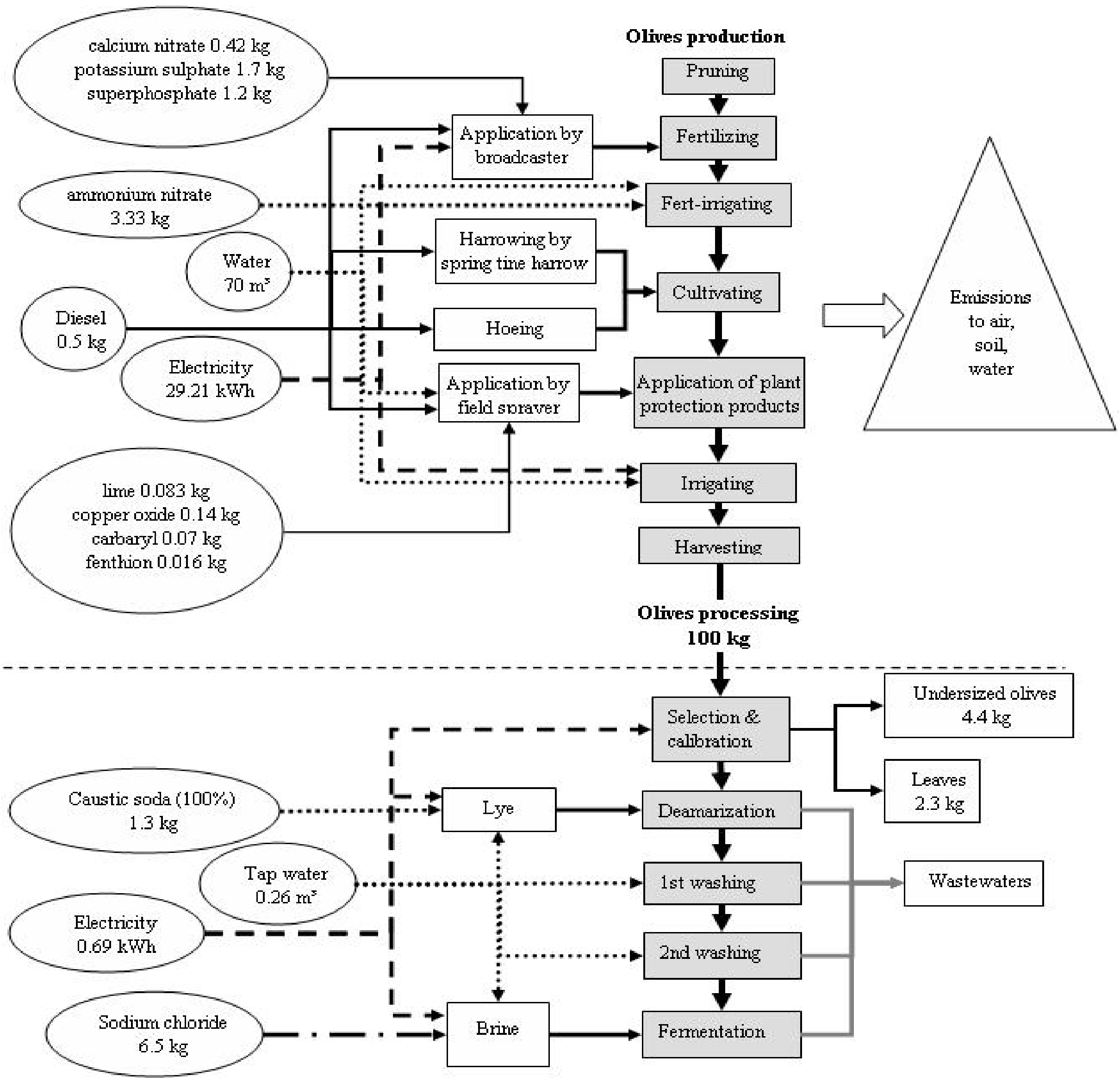


Figure 1: System boundary of the production of Spanish-style green table olives.

3. IMPACT ANALYSIS

For the evaluation the CML2001 method was employed. The categories of environmental impact considered are:

Abiotic Depletion (ADP), Acidification Potential (AP);Eutrophication Potential (EP), Freshwater Aquatic Ecotoxicity Pot. (FAETP inf.), Global Warming Potential (GWP 100 years), Human Toxicity Potential (HTP), Marine Aquatic Ecotoxicity Pot. (MAETP), Ozone Layer Depletion Potential (ODP, steady state), Photochem. Ozone Creation Potential (POCP), Radioactive Radiation (RAD), Terrestric Ecotoxicity Potential (TETP).

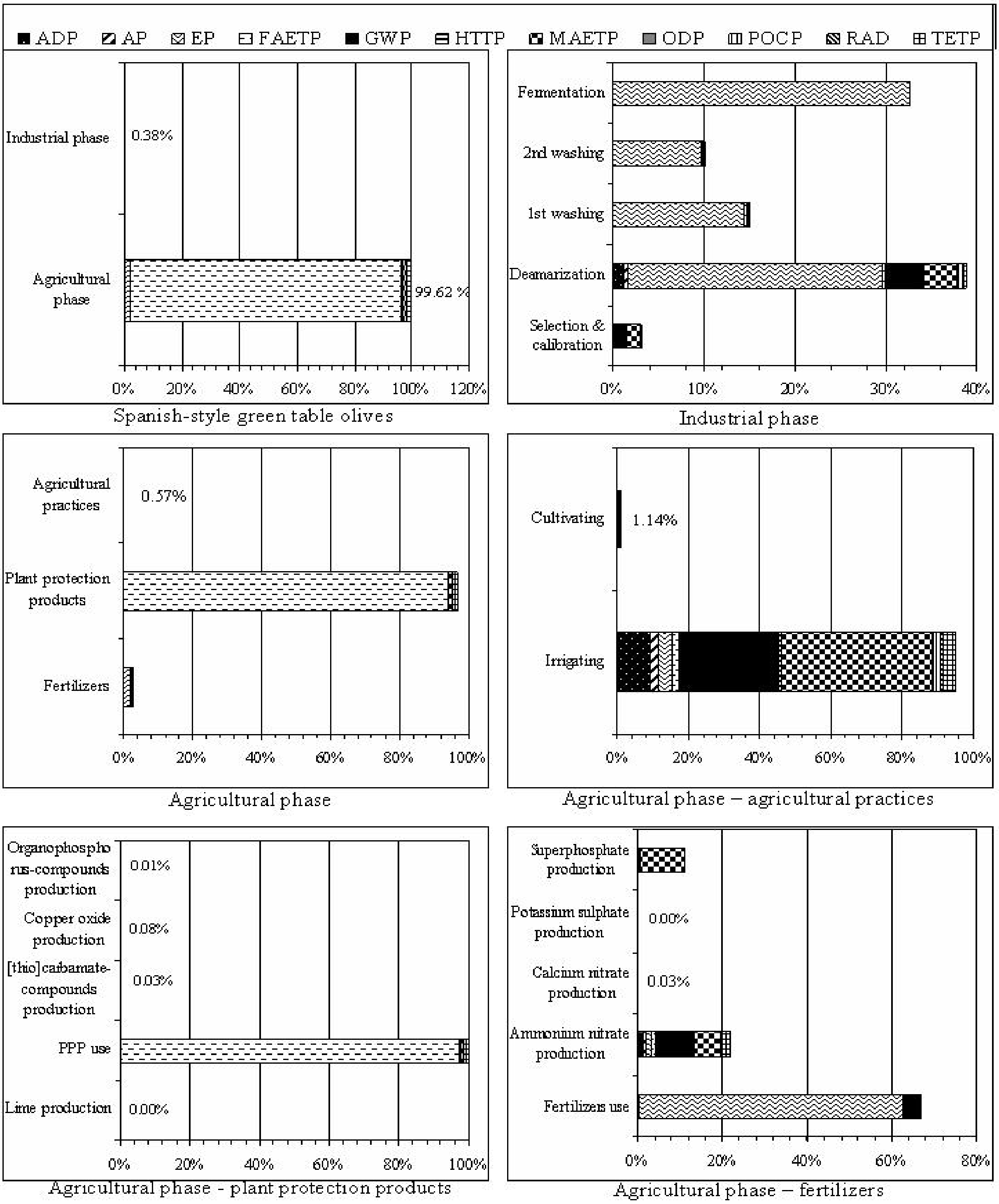


Figure 2: Impact analysis of *Spanish-style* green table olives

The environmental impact of the agricultural phase is substantial when compared to that of the industrial phase.
The extensive use of plant-protection products, to guarantee a crop free from external flaws, has negative effects on the environment. As for cultivation techniques, irrigation is the most polluting. The small quantity of diesel used to power the machines for working the soil and applying fertilizers and plant-protection products has much less impact on global warming than the electricity consumption which is required for pumping copious volumes of water. The environmental pollution resulting from processing the olives by the Spanish method has mainly to do with the waste-waters. The problem concerns the high values of BOD_5 and COD in the lye, the rinsing-water and the fermentation-brine. The eutrophication potential represents the main environmental impact of the industrial phase. The production of caustic soda, which is used for sweetening the olives, further increases the amount of environmental pollution.

4. CONCLUSIONS

As it is the agricultural practices which are mostly responsible for the environmental pollution, these should be carried out in a sustainable way. The use of manure rather than mineral fertilizers would reduce the impact on the environment from the production and application of ammonium nitrate. Similarly, it should be possible to limit the use of plant protection products by favouring crop-protection systems which have less impact on the environment. The employment of traps to capture harmful insects en masse could replace the use of pesticides containing organo-phosphorus compounds, which are toxic both for people and for the environment. The rationalization of water-consumption could be brought about by using a localized drip-irrigation method, with drip-sprinklers buried in the earth to reduce water-evaporation from the soil.
The processing phase should not be thought of as particularly polluting. However, some technological solutions could be employed in order to reduce the amount of caustic soda currently used. It is also worth considering reusing the brine and the rinsing-water, as well as extracting that which is commercially useful, such as phenolic substances.

References

1. www.internationaloliveoil.org
2. www.inea.it/otris
3. F. Brentrup, J. Küsters, J. Lammel, Kuhlmann, Hermann
   Methods to estimate on-field nitrogen emissions from crop production as an input to LCA studies in the agricultural sector.
   The International Journal of LCA 5 (6), 349-357, 2000.
4. M. Hauschild
   Estimating pesticide emissions for LCA of agricultural products
   In B. P. Weidema, M. J. C. Meeusen (ed.): Agricultural data for Life Cycle Assessment, II vol., 64 79, Agricultural Economics Research Institute, The Hague, 2000.
5. Garrido Fernandez, M. J Fernandez Diez, M. R. Adams
   Table Olives
   Chapman & Hall, London, 1997.
6. www.gabi-software.com
7. A. Brighigna
   Le Olive da tavola,
   Ed agricole, Bologna, 1998

Life Cycle Assessment of Silage - comparison of Tower silo, Bunker silo and Round-bales

I. Strid[a] and A. Flysjö[b]

[a]Department of Biometry and Engineering, P.O. box 7032, Swedish University of Agricultural Sciences, 750 07 Uppsala, SWEDEN
[b]Department of Environment and Process Engineering, The Swedish Institute for Food and Biotechnology (SIK), Box 5401, 402 29 Gothenburg, SWEDEN

Abstract

The purpose of this study was to investigate the differences in environmental impact from three types of silage used in Swedish dairy production. The silage was studied from ley cultivation, via harvesting and silage making up to and including delivery at the feeding table. The environmental impact categories studied were: energy use, global warming potential (GWP), acidification potential (AP) and eutrophication potential (EP). The results showed only small differences between the studied silage alternatives. Ley cultivation was the dominant source to GWP, AP and EP for all alternatives. For energy use, the subsequent silage handling chain was more important, including a surprisingly high energy use for producing silage agents. A sensitivity analysis indicated that losses can have a larger impact on the environmental outcome of silage production, than streamlining of individual sub-processes.

1. INTRODUCTION

The majority of the environmental impact of Swedish milk production derives from the agricultural part of the life cycle, compared to e.g., transports, dairy processes and packaging (LRF, 2002). Feed choice and feed production methods hence have a potential to be important management tools for improving the environmental profile of milk.

Since a couple of years, a database of lifecycle assessment results for various feedstuffs is under construction at SIK (The Swedish Institute for Food and Biotechnology). Until recently the database only included protein feeds, but will now be complemented also with roughage feeds (different kinds of silage). Roughage feed accounts for approximately 50 % of the dry matter feed intake of Swedish dairy cows, followed by wheat of c:a 23% (Emanuelsson, et al., 2006). The inclusion of silage in the database will enhance the possibility of testing the environmental impact of different complete feed rations for e.g. dairy cows.

2. GOAL AND SCOPE

The purpose of this study was to investigate the environmental impact from production of three types of silage: tower silo, bunker silo and round-bale silage. The results will then be included in the feed-database at SIK (se above).

The study covers background and on-farm activities in the production of silage; starting with ley cultivation, via harvesting and silage making up to and including delivery at the feeding table (see Figure 1).

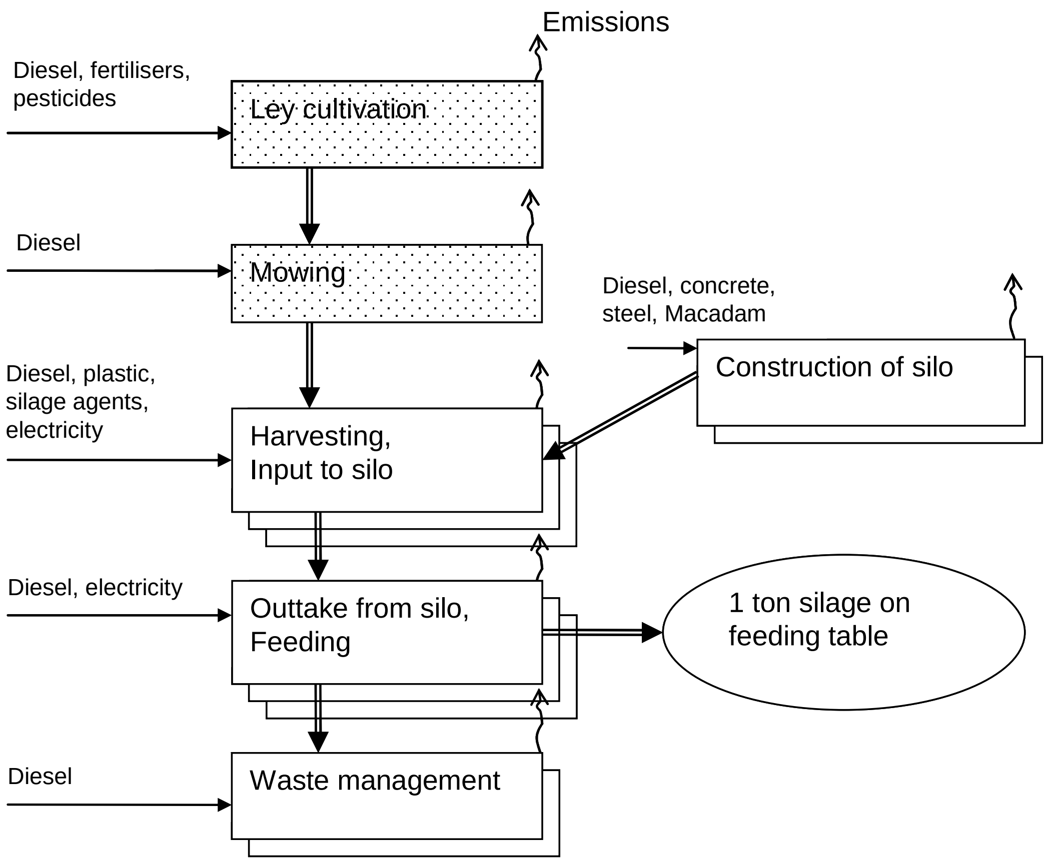


Figure 1. Schematic description of the activities and processes included in the production of silage. The dotted boxes are identical for the three systems.

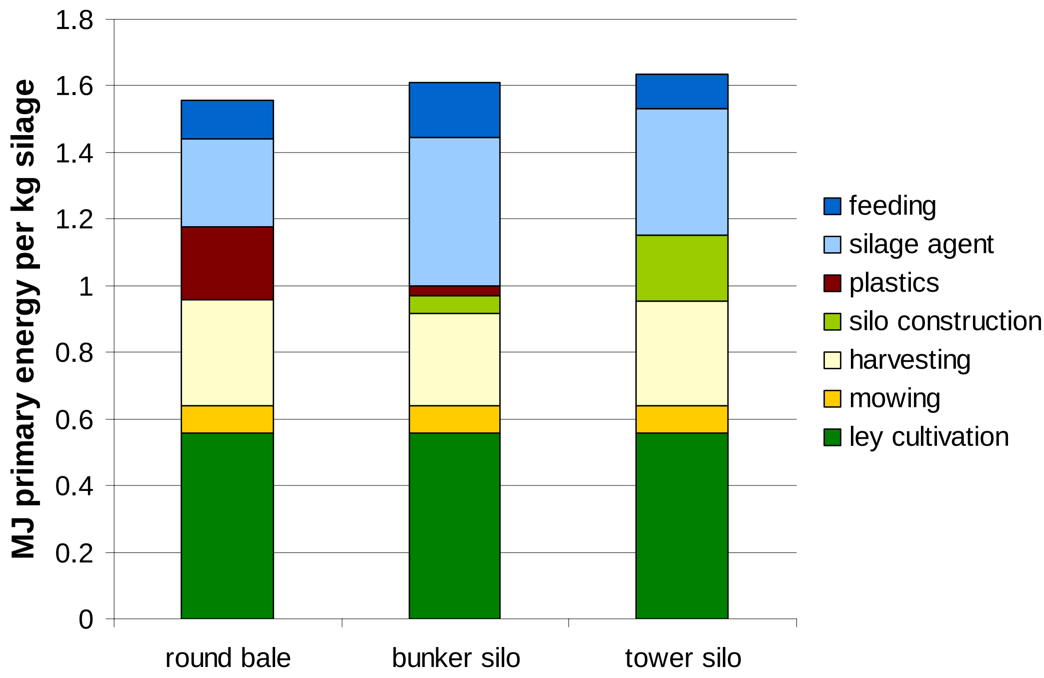
The method used was life cycle assessment and the environmental impact categories studied were energy use, global warming potential (GWP), acidification potential (AP) and eutrophication potential (EP). For calculation of life cycle impacts, the software/database SimaPro 7.0 (Pré, 2004) was used.

The functional unit of the study was 1 kg dry matter silage delivered at the on-farm feeding table.

3. RESULTS AND DISCUSSION

The total environmental impacts of the three studied systems were relatively similar. For GWP, AP and EP ley cultivation was the dominant source (80-98%). For energy use, the subsequent silage handling chain was more important, including a surprisingly high value for producing silage agents (see Figure 2a).
An analysis of the effects of losses were done, where loss factors for round-bale, bunker and tower silage were put to 16%, 13% and 9% respectively (Savoie and Jofriet, 2002). After losses, the absolute energy use increased and the tower silo became slightly more energy efficient than the other alternatives (Fig. 2a and 2b). The analysis also indicated that losses can have a larger impact on the environmental outcome of silage production, than streamlining of individual sub-processes.

E.g., increasing the use of plastics up to 7% would carry a reduced loss of 1% in the round-bale



alternative.
Figure 2a. Energy use to produce one kg silage; no losses included.

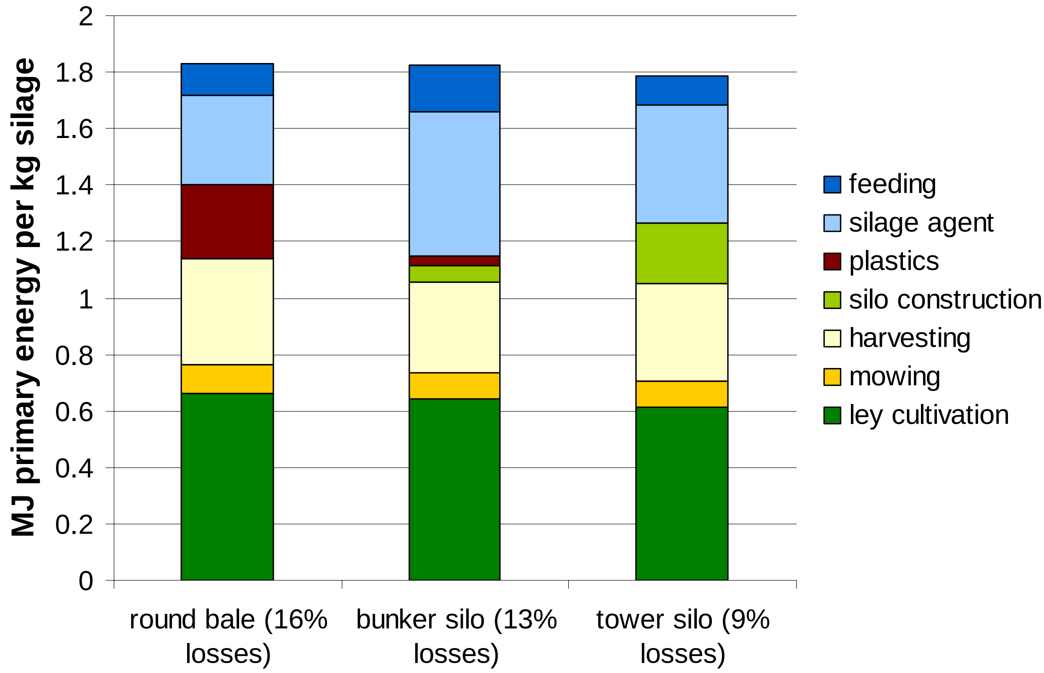


Figure 2b.
Energy use to produce one kg silage; losses (16%, 13% and 9%) included.

If the difference between the studied silage methods does not primarily lie in the amount of resources used for performing alternative managerial processes, such as use of electricity, diesel or plastics, but instead in their potential to produce and preserve large amounts of high quality silage, then the silage method that best succeeds with this task will have the highest potential to be the most environmentally friendly method.

4. CONCLUSIONS

- The differences in environmental impact between the studied silage alternatives were relatively small
- For global warming, acidification and eutrophication ley cultivation was the dominant source (80-98 %)
- For energy use, the silage handling chain was the largest contributor (60%)
- Losses can have a larger impact on the environmental outcome of silage production, than streamlining of individual sub-processes
- The tower silo had the best environmental profile, after losses were included
- The energy use for production of silage agents was surprisingly high
- The silo buildings could lower their environmental burden through longer lifetime, especially the tower silo

5. REFERENCES

Emanuelsson, M., Cederberg, C., Bertilsson, J. Och Rietz, H., 2006. *Närodlat foder till mjölkkor – en kunskapsuppdatering* (*Locally produced feed to dairy cows – update of knowledge*), Report 7059-P, Swedish Dairy Association, Stockholm, Sweden

LRF, 2002. *Maten och Miljön – Livscykelanalys av sju livsmedel* (*Food and the Environment – Life Cycle Assessment of seven foods*, in Swedish), LRF - Federation of Swedish Farmers, Stockholm, Sweden

Pré Consultants bv, 2004, Amersfoort, Holland, www.pre.nl

Savoie, P. och Jofriet, J.C., 2002. Silage Storage in *Silage Science and Technology*, Buxton, D.R., Muck, R.E. och Harrison, J.H. (Eds), 2003. Madison, Wisconsin, USA