PARALLEL SESSIONS 1

1B / Issues in Life Cycle Inventories and datasets

Analysing the influence of the functional unit in agricultural LCA

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ABSTRACT

Impact assessment is largely dependent on the choice of functional unit. In this study we compared two crop alternatives to determine the most appropriate for introduction to the newly irrigated land in Segarra-Garrigues (Spain) using the land area (ha), yield (t) and economic benefit (€) as different bases of comparison. Life Cycle Assessment (LCA) methodology was used to calculate the environmental impacts and Cost-Benefit Analysis (CBA) to determine the economic benefits of each crop alternative. Results showed that horticultural crops would be suitable for the area under study, based on productivity (t) and economic terms (€), because they have higher yield and retail prices in comparison to cereal crops. Further analysis is needed to decide which functional units are the most suitable for agricultural systems, especially when different crops are compared.

Keywords: LCA; Cost-Benefit analysis; Horticultural; Cereal; Corn; Wheat; Onion; Cauliflower.

1. Introduction

Life Cycle Assessment (LCA) is an efficient method to assess agricultural impacts on the environment. Impact assessment is largely dependent on the evaluation objectives, system boundaries and choice of functional unit, and there is a high risk of the environmental assessment being biased by reducing these parameters.

We can use different functional units to express the environmental impacts in agricultural LCA. Currently, the most commonly used are yield (t), land area (ha), nutritional values (e.g. percentage of protein content) and energy content (kJ). Using multiple functional units can improve the interpretation of the environmental results, as mentioned by several authors, such as Haas *et al. (2001)*, Nemecek *et al. (2001)*, Basset-Mens *et al. (2005)* and Charles *et al. (2006)*. In this case study we analyzed the influence of the functional unit in agricultural LCA, and included economic benefit (€) as a common basis of comparison. Authors such as Ross *et al.* (2002) state that the real decision regarding the final choice in product comparisons must be based on LCA, but also on financial and social cost assessments. So this study is framed within the eco-efficiency concept, that is the management philosophy which encourages business to search for environmental improvements that obtain parallel economic benefits.

The main objectives of this study were:

- To analyze and compare the environmental impacts of cereals (wheat and corn) and horticultural crops (onion and cauliflower) using the land area (ha), yield (t) and economic benefit (€) as functional units.

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- To suggest the better crop alternative for introduction in the newly irrigated land in Segarra-Garrigues (Spain) taking into account environmental, productive and economic aspects.
- To analyze the influence of the choice of functional unit in agricultural LCA.

2. Material and methods

2.1. Functional unit

The functional unit is a reference unit against which the inventory data and results are normalized. The use of multiple functional units can improve the interpretation of the environmental results obtained in LCA studies. In this work, the land area (ha), yield (t) and economic benefit (€) were chosen as bases of comparison.

2.2. Systems description and boundaries

The systems analysed were: cereals (corn and wheat) and horticultural crops (onion and green cauliflower). Cereals were chosen as one alternative as they are the most widely cultivated crops in the irrigated land near the area under study. Horticultural crops were chosen since they are gaining popularity in these areas and, furthermore, climate conditions are especially favourable for them. We considered the most common varieties of horticultural crops, taking into account their own characteristic harvest productions, which are slightly lower than the average national value (mainly in the green cauliflower crop).

Information on harvest production (t ha^{-1}), field processes, inputs (seeds ha^{-1}) and fertilizers and pesticide use (kg ha^{-1}) was from local studies (Seda *et al.*, 2010) and national studies (Porcuna, 2007; Macua *et al.*, 2009). The average retail prices (€ t^{-1}) for the period 1995-2008, in the province of Lleida (Table 1), was from local sources (DAR, 2009).

Table 1. Overview of the agronomic production data for corn, wheat, onion and green cauliflower in the area under study.

	Crop	Corn	Wheat	Onion	Green cauliflower
Outputs	Harvest (t ha^{-1})	13	6	60	16
	Retail prices (€ t^{-1})	152	152	234	366
Inputs	Sowing density (seeds ha^{-1})	95,000	400	650,000	22,000
	Fertilizer use [(3)]				
	Complex NPK (kg ha^{-1})	100	100	0	0
	Potassium nitrate (kg ha^{-1})	1,184	308	227	340
	Monoammonium nitrate (kg ha^{-1})	0	0	164	408
	Monoammonium phosphate (kg ha^{-1})	0	0	131	163

The system boundary was set at the farm gate because the main goal was to study the agrarian production system. Both for LCA and for cost-benefit analysis, field preparation, fertilization, sowing (or planting for horticultural systems), phytosanitary treatments, irrigation system and harvesting processes were included. Labour was only included in the cost-benefit analysis. It could be interesting to develop in a future a more complex LCA including social aspects or maybe doing a consequential LCA.

3. Life Cycle Inventory analysis (LCI)

SimaPro v.7.1 software was used for the analysis of impacts, only performing the obligatory classification and characterization.

The proportional fraction of machinery used in each field operation was included in order to calculate the material required in each one. Generalized and standard production processes were taken from Nemecek *et al.* (2007), and emissions to the environment from Nemecek *et al.* (2007) and Martínez Gasol (2006).

Nitrogen, phosphorus and potassium application rates were based on local recommendations. We assumed an NH_3 volatilization factor of 3% (Audsley, 1997), an N_2O emission factor of 1.25% of N addition (Brentrup *et al.*, 2000) and NO_x emissions were calculated as 10% of the N_2O emissions (Audsley, 1997). We assumed that nitrate leaching was negligible because an adequate fertilizer application rate and an appropriate irrigation plan were developed (Doltra *et al.*, 2010).

In this study we did not applied any method of allocation, and all the environmental impacts were related to the cereal or horticultural crops.

4. Cost-benefit analysis

In this study we used a simply cost-benefit analysis to estimate costs and profits. Yield (t ha^{-1}) and harvest retail prices (€ t^{-1}) were used for calculating profit (€ ha^{-1}) for each crop alternative (see Table 1), and costs of machinery, inputs (fertilizers, seeds, phytosanitary products, etc.) and labour for calculating the total cost (€/ha). Finally, the benefit obtained was calculated by subtracting the total cost from the profit for each crop alternative.

To calculate the machinery costs, we considered interest cost, depreciation cost, repair and maintenance costs and fuel cost (ASABE D497.4, 2003).

5. Results and discussion

The impact categories assessed in this study were abiotic depletion (AD), global warming potential (GWP), ozone layer depletion (OLD), photochemical oxidation (PO), air acidification (AA) and eutrophication (EU), the most commonly used in LCA. We did not consider impact categories related to human and ecosystem toxicity due to the lack of scientific consensus on calculation methods. Table 2 shows the life cycle impacts produced by the two analysed systems related to land area (ha), yield (t) and economic benefit (€).

Table 2. Life cycle impacts produced by cereal and horticultural crop alternatives related to land area (ha), yield (t) and economic benefit (€).

	Environmental impacts/ha		**Environmental impacts/t**		**Environmental impacts/€**	
Impact categories	**Cereal**	**Horticultural**	**Cereal**	**Horticultural**	**Cereal**	**Horticultural**
AD (kg Sb eq)	2.87E+01	3.52E+01	1.51E+00	4.63E-01	1.65E-02	2.85E-03
GWP (kg CO2 eq)	6.60E+03	7.07E+03	3.48E+02	9.31E+01	3.79E+00	5.73E-01
OLD (kg CFC-11 eq)	5.50E-04	5.30E-04	2.89E-05	6.97E-06	3.16E-07	4.29E-08
PO (kg C2H4)	9.80E-01	1.37E+00	5.16E-02	1.80E-02	5.63E-04	1.11E-04
AA (kg SO2 eq)	3.75E+01	4.57E+01	1.97E+00	6.01E-01	2.15E-02	3.70E-03
EU (kg PO4-3 eq)	5.31E+00	1.58E+01	2.79E-01	2.08E-01	3.05E-03	1.28E-03

5.1. Comparison per hectare (ha)

In the first approach, the assessment was based on the area of crop production, and impacts were given per hectare (1 ha). This functional unit provides explicit information on the intensity of use of agricultural inputs. The results show that the impacts from horticultural crops were higher than from cereal crops in five of the six environmental categories considered, using 1 ha as the functional unit (Table 2). The greatest differences were in EU, followed by PO, AD and AA.

Different values for horticultural and cereal crops in the EU category could be attributed to different doses and types of P-fertilizer applied in each case. Different values in the PO category could be attributed to the differences between the crops, the harvest machinery employed and the number of repeat passes required for each one. In the AD category, natural gas consumed during N fertilizer production was the major contributor. Finally, emissions of ammonia (NH_3) during application of mineral fertilizers were the most significant emissions in the AA category.

5.2. Comparison per ton of product (t)

In the second approach, the assessment was based on production, and the environmental impacts were expressed per ton of product obtained (t). This is a reflection of agricultural activity as a producer of market goods, and it can be used to evaluate the effect of cultivation techniques on yield (e.g. different rates of fertilization).

We found that, in the Segarra-Garrigues area, the horticultural crops produced lower impacts per t than the cereal crops in all environmental categories considered (Table 2), with the greatest differences in OLD, followed by GWP, AD and AA. This is largely attributable to the higher yields with horticultural crops, making them a good candidate for commercial production, based on yield (see Table 1). Better environmental performance for cereal crops production could be achieved by improving yields with more efficient land use.

5.3. Comparison per economic benefit obtained (€)

In the third approach, the assessment was based on the economic purpose and the functional unit was 1 € of economic benefit. This assessment is a helpful tool for decision makers to judge the significance of the differences in product comparisons.

Table 2 shows that horticultural crops could be considered a suitable choice based on economical terms, mainly due to higher retail prices and yields. These crops produce lower impacts per € of benefit than the cereal crops in all the environmental categories considered, with greatest differences in OLD, followed by GWP and AA.

The cost-benefit analysis revealed that the economic benefit of the horticultural crop alternative was 7 times higher than cereals (12,344 and 1,741 €/ha, respectively). The horticultural system has higher costs than cereals (7,552 and 1,147 €/ha, respectively) but also higher profits (19,896 and 2,888 €/ha, respectively). The main costs for horticultural crops were assigned to labour mainly in harvesting operations. In contrast, the main costs for cereals were related to inputs and machinery.

5.4. All comparisons

Figure 1 shows the different cereal/horticultural ratios for each functional unit and impact category. These ratios were obtained by dividing the environmental impacts of cereals by those of horticultural crops, expressed per hectare, per ton or per euro.

When ha of land was used as the basis for comparison, ratios obtained were < 1 in five of six impact categories, with values between 0.34 and 0.93. This means that horticultural crops produced more impact than corn and wheat. Horticultural crops have high nutrient requirements due to intensive management, involving high fertilizer application rates and environmental impacts.

In contrast, when weight of product or economic benefit were used as functional units, the ratios obtained were > 1 in all of the impact categories. In the first case, the values obtained were between 1.35 and 4.15, with the highest values in the OLD and GWP categories. In the second, these ratios were between 2.39 and 7.36. This mean that cereal crops produced more impact than horticultural crops when t or € were used as functional units. These differences could be attributed to higher yield and retail prices of horticultural crops in comparison to cereals.

The uncertainty of outputs (such as retail prices or yield of horticultural crops) and inputs (such as fertilizers or pesticides use) also need to be taken into account in further studies. The retail prices (€ t^{-1}) considered in this study represented the average over the period 1995-2008 in the province of Lleida, but it is known that this could vary depending on the year. Data on yield (t ha^{-1}) for horticultural crops was taken from local studies in nearby areas over approximately 4 years; using national average values could give correct statistical results but would not be realistic for this area. With more local data on yield, it could be interesting to consider its variability. Data on fertilizers (kg ha^{-1}) was taken from local recommendations and its uncertainty could be considered due to the inherent variability of agricultural processes.

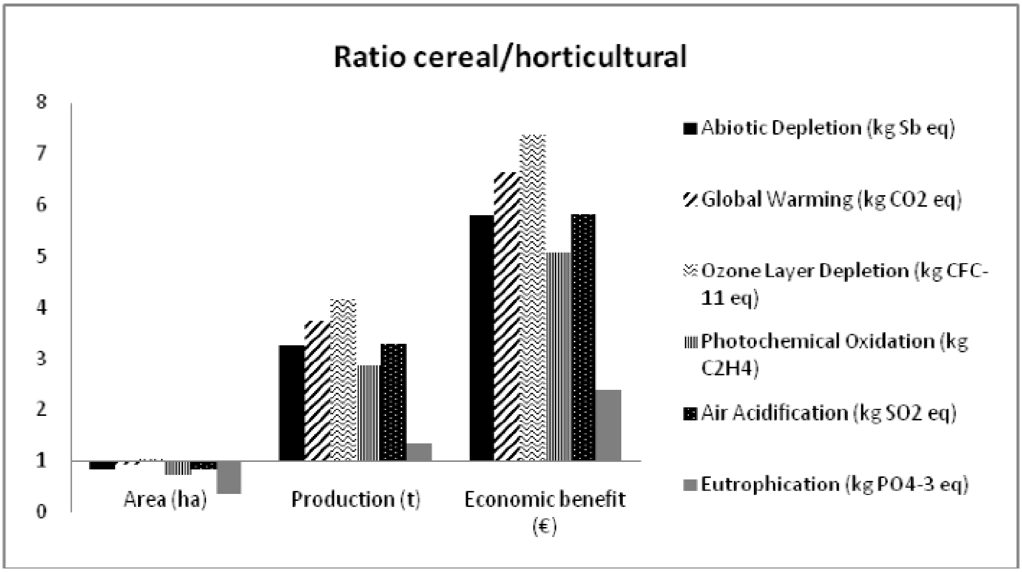

Figure 1. Ratio cereal/horticultural in six environmental categories using area (ha), yield (t) and economic benefit (€).Values > 1 indicate higher environmental impact for the cereal than horticultural crop alternative. Values < 1 indicate more environmental impact for the horticultural than cereal crop alternative.

6. Conclusions

LCA methodology is very useful to evaluate environmental impacts of a product or production system. A comparison of two agricultural systems cannot be reduced to an environmental analysis alone, and introduction of economic and other approaches would provide much more reliable and comprehensive information for policy makers and producers to select sustainable products.

From this case study it can be concluded that horticultural crops would be a suitable choice based on productivity and economic terms. The differences could be attributed to higher yield and retail prices of horticultural crops in comparison to cereals. Further analysis is needed to decide which functions are the most suitable for agricultural systems, especially when different crops are compared. Future study should focus on including social aspects in LCA.

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Using Life Cycle Inventory systems modelling to determine the limits to sustainable livestock production

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ABSTRACT

The systems model Life Cycle Analysis (LCA) approach combined with linear programming (LP) can determine the size and configuration of the livestock industry which optimally meets the UK's future emission targets. The LCA quantifies inherent differences in emissions between the many current production systems within each livestock sector, estimates the impacts of abatement techniques on emissions and calculates the demand for other resources such as feeds and land. The LP maximises output which meets reduced emission targets, subject to constraints between systems, land types available and levels of production. Allowing production of every sector to fall to 72% of current, the best that can be achieved is 87% of overall current production by value, 93% by protein and 95% by energy. 92% value is reached with 10% increase in FCR. It is easy to envisage systems that maintain current production within NH_3 limits but impossible for GHG limits.

Keywords: agriculture, livestock, food consumption, system modelling, emission targets,

1. Introduction

The UK has made formal commitments to reduce ammonia (NH_3) and greenhouse gas (GHG) emissions from agriculture. The majority of these emissions are due to livestock, either directly, from their feed production or manure management. Dairy cows, beef and sheep are major methane producers, whereas pigs, poultry and eggs are mainly ammonia producers. What configuration of the livestock industry can best meet the UK's commitments? It is important to remember that there are additional commitments such as to not export the greenhouse gas production of our food consumption and to promote sustainability in terms of economic productivity and food security. Thus given the commitments, which would systems maximise food production: hill or lowland sheep, indoor or outdoor pigs and poultry, slurry or farm yard manure waste handling, intensive cereal or extensive grass-fed beef, higher yielding dairy cows, etc? What are the improvements to the systems that would help?

2. The model

The LCA procedure (Williams et al 2006, Audsley and Williams, 2008) calculates the emissions for each sub-system within each sector (eg silage-fed 18-month beef production, using calves from the dairy herd, with slurry manure handling), thereby producing a set of inventories for each sub-system within each livestock sector (sheep, beef, dairy, eggs, poultry meat, pig meat). The land used is also calculated and divided into arable and hill, upland and lowland grassland. The systems modelling approach to the LCA also enables calculation of the effect of proposed improvements such as actions to reduce emissions (slurry injec-

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tion), higher yielding dairy cows, or improved food conversion efficiency, such as in pigs and poultry.

The above 'what configurations' question can be expressed in a mathematically simple way as - find the values of x_{ij} such that:

$$\text{maximise} \sum v_{ij}x_{ij} \text{ such that} \sum g_{ij}x_{ij} \leq G \text{ and} \sum a_{ij}x_{ij} \leq A$$

Where i is the commodity, j the subsystem and v, g, a are the output, global warming potential and ammonia emission respectively, of each subsystem. In other words this is a linear programme (LP) which determines the combination of systems which maximises output whilst achieving the objectives of a reduced level of total emissions. Further constraints need to be added to this simple LP in order to properly describe the systems of production. Figure 1 shows a very simplified version of the LP with the sub-systems eliminated to illustrate the concept. The full model contains 211 columns.

	Pig	Poultry	Beef	Sheep	Dairy	Layers	CO_2	CH_4	N_2O		RHS
OBJ	1.55	0.90	2.27	2.38	2.53	0.83	0	0	0		
CO2k	2.55	1.83	3.21	3.13	2.50	1.98	-1	0	0	=	0
CH4k	0.06	0.00	0.33	0.35	0.21	0.01	0	-1	0	=	0
N2O	2.88	3.30	12.59	14.25	6.22	3.56	0	0	-1	=	0
sumGHG	0.00	0.00	0.00	0.00	0.00	0.00	1	23	0.296	<	22.15
NH3	36.72	17.69	70.44	27.77	40.52	29.37	0	0	0	<	109.48
Pig_prod	1	0	0	0	0	0	0	0	0	>	0.45
Poul_prod	0	1	0	0	0	0	0	0	0	>	1.03
Beef_prod	0	0	1	0	0	0	0	0	0	<	0.44
Shp_prod	0	0	0	1	0	0	0	0	0	<	0.22
Dairy_prod	0	0	0	0	1	0	0	0	0	>	0.89
Eggs_prod	0	0	0	0	0	1	0	0	0	>	0.30
LU_G1max	0.00	0.00	0.00	0.00	0.03	0.00	0	0	0	<	0.07
LU_G2max	0.00	0.00	0.05	0.08	0.21	0.00	0	0	0	<	0.68
LU_G3max	0.00	0.00	0.09	0.26	0.40	0.00	0	0	0	<	2.98
LU_G3amax	0.00	0.00	0.82	1.44	0.00	0.00	0	0	0	<	2.71
LU_G4max	0.00	0.00	0.66	1.16	0.00	0.00	0	0	0	<	2.36
LU_G5max	0.00	0.00	1.41	2.87	0.00	0.00	0	0	0	<	1.26
Grassland	0.00	0.00	3.03	5.82	0.65	0.00	0	0	0	>	0.65
Total_land	0.69	0.62	3.58	6.08	1.03	0.57	0	0	0	<	10.06
LU_G1<100	0.00	0.00	0.00	0.00	0.00	0.00	0	0	0	>	0.00
LU_G2<100	0.00	0.00	0.05	0.08	0.00	0.00	0	0	0	>	0.01
LU_G3<100	0.00	0.00	0.09	0.26	0.00	0.00	0	0	0	>	0.03
Beefdairytie	0.00	0.00	2.02	0.00	-0.80	0.00	0	0	0	=	0.00

Figure 1: Illustration of the livestock configuration LP model showing commodity level only

The subsystems are not independent of each other. Figure 2 shows the structure of the beef industry as an example. Calves can come from the beef or dairy cows, there is a known proportion of male and female calves (some female calves are required to replace the cows), calves can be fattened intensively or extensively, and there are differences in the growth rate and killing out percentage from the different breeds/sexes. Each link between the systems is described within the LP by a constraint. The same procedure applies to each commodity. Thus for the pig sector: indoor or outdoor sows produce piglets, which become indoor or outdoor weaners which become light, medium or heavy finishers. Indoor can be either slurry or straw-based. In addition for beef and milk, there is a constraint between commodities since dairy cows are required for there to be (surplus) dairy-bred calves for beef, which means that dairy farming implies a minimum level of beef production.

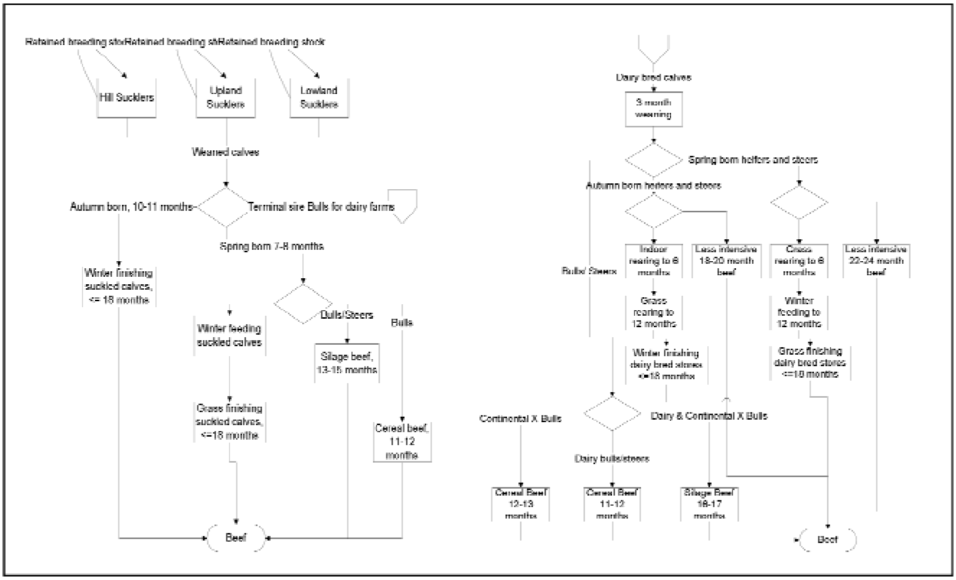

Figure 2: Beef structural system showing links between sub-systems

One can also impose production restrictions on each commodity. Thus one can require a minimum level of production from each sector, or limit production to some level – for example the current demand. Although not used in this study, it is also possible to impose constraints on the timing of production, such as no more than 60% of the milk is produced in either autumn or spring. Typically one would expect milk produced from grass to be more efficient than that produced from silage. These timing constraints however, need to link with the possibilities of imports – the most notable example being New Zealand lamb in the spring when British lamb is in short supply. Figure 3 illustrates the effect on supply of lowland ewes which produce finished lambs in summer versus hill ewes which produce finished lambs in winter.

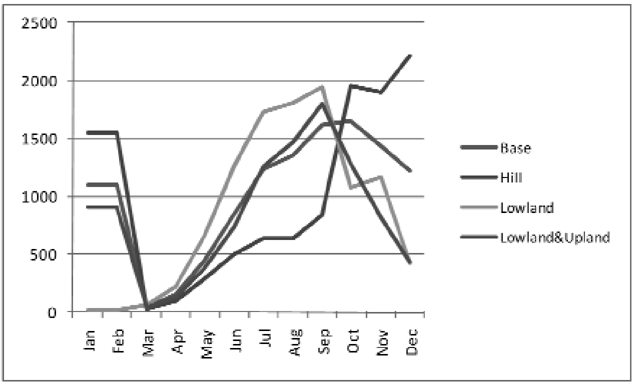

Figure 3: Production of finished lamb using hill, upland and lowland ewes

Finally the solutions require a quantity of hill and upland land, which are of course restricted to the present amounts, even if hill systems are the best at reducing emissions. Thus constraints are imposed on these areas of land. Lowland and arable land are not constrained, though clearly there would be implications for other lowland production, and constraints could be added if required.

The measure of the value of the output from each sector is a cause for concern. The major output of livestock system is protein, thus value can be the protein content. However whereas meat contains only protein and fat, milk and eggs contain carbohydrates. Thus an alternative value measure is energy. Similarly red meat contains key vitamins not present in white meat. Thus another alternative is to use the monetary value of the output at the farm gate to express the combined value of the nutrients in the products. In turn however value expresses supply-demand and health scares – for example the price of eggs have increased since it was recently declared to not be bad for your health! We therefore solve for all three measures and compare the results. In the majority of cases, the results are similar.

The first step is to solve the model for the current level of demand for each commodity and the current make-up of the sub-systems. To do this more constraints are added expressing the proportion of each of sub-systems – for example 58% of indoor sows are slurry-based, and 37% of ewes are on the lowlands.

3. Results

The results are all described for a 20% reduction in the emissions of GWP100 and Ammonia.

Analysis of each livestock sector in turn with the existing production systems indicates the optimum reconfiguration of each sector. Reducing gaseous emissions by 20% would enable:

- 84% of sheep production by value using fewer hill and more lowland and no early lambing.
- 88% of pig production by value using outdoor sows and weaners with medium weight finishing with slurry. NH_3 is not limiting.
- 88% of beef and milk production by value using spring calving low yielding with slurry and with a few Autumn low yielding with FYM to balance NH_3. No sucklers are used. Note however that milk production alone is optimal with high yielding spring calving cows but fewer beef calves are then produced.
- 81% of poultry production using housed, rather than free-range.
- 82% of egg production using housed, rather than free-range.

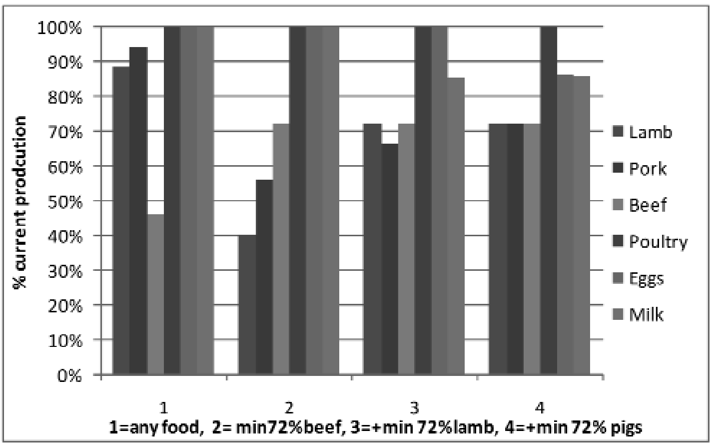

Figure 4: Optimise value of food production with 20% reduction on GWP and NH_3

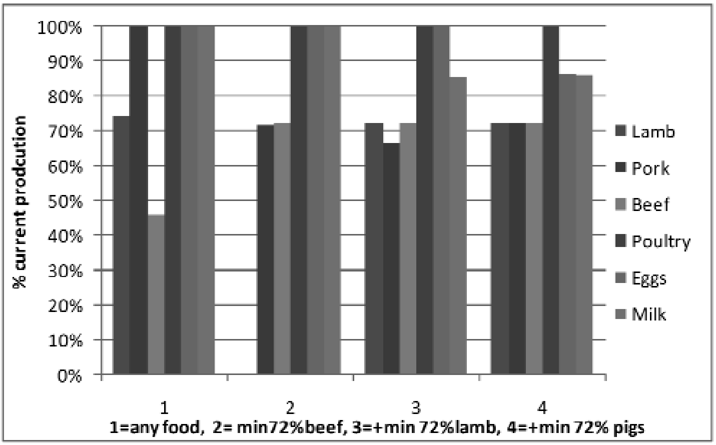

Figure 5: Optimise protein of food production with 20% reduction on GWP and NH_3

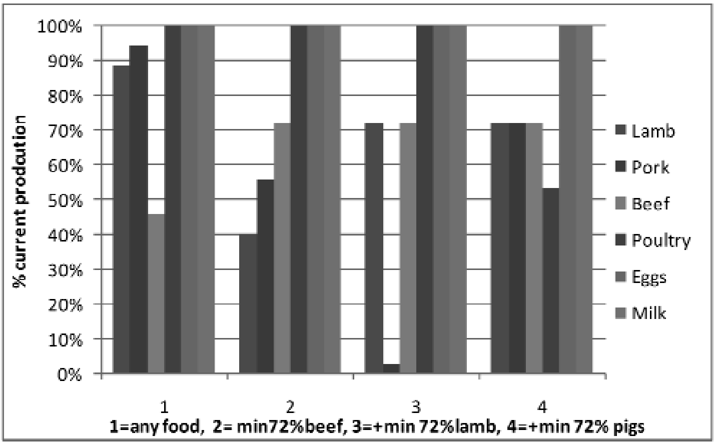

Figure 6: Optimise energy of food production with 20% reduction on GWP and NH_3

Results combining all the sectors and allowing production of every sector to fall to 72% of current levels, suggest the best that can be achieved is 87% of overall current production by value, 93% by protein and 95% by energy. However the higher energy solution is the same as the value solution because beef and lamb are a low proportion of the total energy. Figs 4-6 show the differences between optimising by value, protein and energy.

- By value, suckler beef is removed and there is some reduction in lamb and pork. Requiring 72% beef production limits lamb and pork. Requiring 72% lamb reduces milk and eggs. Requiring pork reduces eggs and milk.
- By protein, suckler beef and only lamb are reduced. Requiring 72% beef production eliminates lamb and reduces pork. Requiring 72% lamb reduces milk. Requiring pork reduces eggs and milk.
- By energy, there is less reduction in lamb and pork. Requiring 72% beef production causes a large reduction in lamb and pork. Requiring 72% lamb almost eliminates pork. Requiring pork reduces poultry.

Thus there are broad similarities but differences.

Diets to reduce rumen fermentation which reduces methane production would enable 0.5% additional milk production. Increasing Food Conversion Ratio for all the systems increases the total value possible from 87% to 90%, but in this case the limiting factor is ammonia not GWP. Allowing the ammonia limit to be breached permits 92% of value, and all current demands can be met except beef which is at 60%.

Most solutions use less than the available amount of hill land, for which there is very little alternative use. Imposing the requirement to use this hill land, only allows 81% of value to be achieved, although 105% of current lamb demand is produced. An alternative to use up the land is to switch to organic sheep. This allows 96% of the overall livestock value to be achieved but only produces 57% of the current lamb demand. This illustrates that lamb has a low value versus its production of greenhouse and ammonia gases.

Allowing the optimal choice of FYM versus slurry produces quite a complex set of choices which are trying to match savings in ammonia and GHG emissions. For pigs, breeding and fattening should use slurry, whilst weaners should use FYM. Beef systems are even more mixed, with the optimum including that 18-month male beef should use slurry but female should use FYM. Dairy cows should use slurry. Two points should be made about these results. Firstly the emission rates of ammonia are taken directly from the national inventory and some such as weaners appear odd. Secondly where the emission rates are very similar, the LP will still choose the best, whereas the alternative might be negligibly different.

The model also considered numerous methods of reducing ammonia emissions from livestock waste. These included covering slurry stores, ploughing immediately after spreading, injecting slurry, drying poultry waste, and so on. Some of these solutions, whilst reducing ammonia, actually increase GWP because of the energy required to implement them. Thus in the majority of cases, the result was that solutions were limited solely by the GWP and the ammonia limit was no longer a constraint. A major conclusion is that it is easy to envisage systems that exceed the reductions required in NH_3 emissions whilst maintaining current levels of production, albeit the cost of implementation might be considerable. It is impossible to envisage systems which maintain current production levels and meet the GHG limit.

A corollary to this analysis, is that in order to reduce emissions one needs to reduce the amount of livestock products in the diet. The subject of Williams et al. (2010).

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Modular extrapolation of crop LCA (MEXALCA): Sensitivity to varying crop yields

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ABSTRACT

A method for the geographical extrapolation of farming inputs and environmental impacts, MEXALCA, was investigated with respect to its sensitivity to variations of crop yields as evident from public statistics. A case study on wheat revealed an increase of the average global yield from 2300 (1983–1987) to 2820 kg wheat ha^{-1} (2003–2007, today's conditions) to be reflected in a 19 % average rise of the global warming potential (GWP) and the non–renewable energy demand per hectare. The corresponding impacts per kilogram wheat decreased by 10 %. Comparison of today's conditions with an average global yield of 2580 kg ha^{-1} (1993–1997) leads to 11 % (GWP) or 9 % (non–renewable energy demand) higher impacts per hectare, while the generic impacts per kilogram remain at about the same average value. The analysis revealed a strong dependency of the extrapolated inputs or impacts on the yields given for the original country.

Keywords: geographical extrapolation, variability, life cycle inventory, crop LCA, wheat

1. Introduction

Life Cycle Assessments (LCAs) are increasingly used in the food sector to estimate the environmental impacts of agricultural and processed products. However, data on such diverse production systems are seldom available and it is too time and cost intensive to calculate detailed LCAs for a multitude of products and ingredients originating from all over the world. In order to overcome this problem, several approaches are currently applied, e.g. the use of proxy data and generalisations (Muñoz *et al.,* 2010) or simplified LCAs that do not consider all processes involved (Kuan *et al.*, 2007, Zah *et al.*, 2009).

This study investigates a third approach, which is the geographical extrapolation method proposed by Roches *et al.* (2010), aiming at a simplified assessment for agricultural and horticultural crops for all producing countries worldwide, while still considering all relevant processes. MEXALCA (Modular EXtraplolation of Agricultural Life Cycle Assessment) is based on the assumption that the environmental profile of agricultural systems can be described by nine key farming operations (Nemecek *et al.*, 2005, Roches *et al.*, 2010) named modules. These are basic cropping operations, tillage machinery use, variable machinery, nitrogen, phosphorus and potassium fertiliser use, pesticide use, irrigation and drying.

A detailed Life Cycle Inventory (LCI) for a crop in a country (original country) is extrapolated to another (target) country by scaling the inputs induced by each of the modules. For scaling, estimators depending on the ratio of the yields and the farming intensities (agricultural indices) in the target and original countries are defined (see section 2). Both crop yields and farming intensities are country specific, however the latter are not crop specific but represent the prevailing economic situation or traditions specific to a certain country. Both factors are derived from FAO statistics (FAO, 2010) and EarthTrends (WRI, 2009). A list of the agricultural indices used is given in Roches *et al.* (2010). Based on the extrapolated LCIs the Life Cycle Impact Assessment (LCIA) of the crop in the target countries is

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derived. The same characterization factors are applied to all countries. As from now the extrapolation results are referred to as generic data.

A first validation of the generic LCIA results showed MEXALCA to perform well for the impact categories global warming potential (GWP), non–renewable energy demand, and photochemical ozone formation (Roches *et al.*, 2010). Data generated with MEXALCA are not intended to replace LCIA studies based on detailed input data sets referring to a specific crop production in a certain country; rather, they are meant to inform strategic decision making, identify hot spots of environmental impacts during the crop production stage, and help to understand the geographical variability of production systems on large spatial scales.

This paper addresses the sensitivity of the MEXALCA model to the variation in crop yields over the last three decades that is evident from FAO statistics (FAO, 2010). Crop yields reflect the technological and economic development of a country. At the same time, they also depend on political regulations or the occurrence of natural disasters. In a case study on wheat production up to the farm gate the effects of changing yields on the average generic inputs and impacts for two functional units, per hectare and per kilogram of wheat, are investigated. The extrapolation is based on the LCI of wheat at farm in Switzerland (Roches *et al.*, 2010).

2. Calculation of the generic inputs and impacts using MEXALCA

In order to extrapolate the original country inputs and to derive the corresponding impacts for all other wheat producing countries, estimators are defined for each of the nine modules (Roches *et al.*, 2010). The yield ratio, i.e. the yield in the target country divided by the yield in the original country, explicitly occurs in the estimators for the modules variable machinery use, nitrogen, phosphorus and potassium fertilizer use and pesticide use:

$$\hat{X}_t^c = X_o^c \cdot \frac{Y_t^c}{Y_o^c} \cdot \sqrt{\frac{ind_t^X}{ind_o^X}}. \tag{1}$$

In addition, the yield ratio is used in the estimator for the module drying:

$$\hat{X}_t^c = X_o^c \cdot \frac{Y_t^c}{Y_o^c} \cdot \frac{ind_t^X}{ind_o^X}. \tag{2}$$

$\hat{X}_t^c$ or X_0^c are the amounts of farming input in the target (subscript *t*) and original (subscript *o*) country, respectively for production of crop *c* (intensity index for variable machinery use, kg N ha^{-1}, kg P_2O_5 ha^{-1}, kg K_2O ha^{-1}, kg active ingredient ha^{-1}). Y_t^c and Y_o^c are the yields in the target and original countries (kg raw product ha^{-1}), and ind_t^X and ind_o^X are the agricultural indices in the target and original countries, respectively, representing the intensity of input use (Roches *et al.*, 2010).

The estimators for basic cropping operations, tillage machinery use and water use do not include the yield ratio.

3. Variation of yields

3.1. Input scenarios and statistical measure

In order to study the sensitivity of the generic inputs and impacts to changing yields, 5–year averages are calculated using country specific wheat yields (FAO, 2010) and three different scenarios as an input to the MEXALCA model: 1983–1987 (scenario 1), 1993–1997 (scenario 2) and 2003–2007 (reference scenario reflecting today's conditions). Globally, average wheat yields increased during these three time intervals from 2300 kg ha^{-1} (scenario 1) to 2580 kg ha^{-1} (scenario 2) and 2820 kg ha^{-1} for the reference scenario. The reference scenario is indicated with a subscript *ref*, while the other intervals are marked with a subscript *int*.

Weighted averages of the generic farming inputs and environmental impacts with respect to the different yield scenarios are used as a measure for comparison. The contribution of a country to the total world production (FAO, 2010) during the time intervals mentioned above is applied as a weight. In Figures 1 and 2, farming inputs and environmental impacts are depicted with respect to the cumulated world production (in %). This is the summation of each country's contribution to the world production of wheat, while the values are sorted in ascending order on the y–axis, i.e. the generic farming inputs and environmental impacts.

3.2. Generic farming inputs per hectare

Based on the assumption that the farming inputs per hectare $\hat{X}^c_{t,\mathrm{int}}\big|_{\mathrm{ha}}$ are linearly related to the yield ratio (see equations 1 and 2), the application of a different yield scenario leads to the following expression for the $\hat{X}^c_{t,\mathrm{int}}\big|_{\mathrm{ha}}$ with respect to the reference $\hat{X}^c_{t,ref}\big|_{\mathrm{ha}}$:

$$\hat{X}^c_{t,\mathrm{int}}\Big|_{\mathrm{ha}} = \hat{X}^c_{t,ref}\Big|_{\mathrm{ha}} \cdot \left(\frac{Y^c_t}{Y^c_o}\right)_{\mathrm{int}} \Bigg/ \left(\frac{Y^c_t}{Y^c_o}\right)_{ref} . \tag{3}$$

Thus, for a yield ratio $\left(Y^c_t / Y^c_o\right)_{\mathrm{int}}$ that is larger (or smaller) than the reference yield ratio $\left(Y^c_t / Y^c_o\right)_{ref}$, higher (or lower) generic farming inputs per hectare will result.

Farming inputs per hectare wheat with respect to the cumulated world production (in %) are exemplarily shown for the amount of nitrogen (N) fertilizer input (Fig. 1a) and drying input (Fig. 1b, expressed as the amount of water extracted, Roches *et al.*, 2010). Each step represents the generic farming input of a country based on the different yield scenarios. In agreement with the temporally increasing mean wheat yields (section 3.1) and as expected from equation 3, farming inputs per hectare resulting for scenarios 1 and 2 (blue and green lines) are generally lower than the reference scenario (red lines) representing today's conditions.

3.3. Generic farming inputs per kilogram of product

The generic farming inputs per kilogram of product are calculated from those per hectare by dividing by the yields in the target country $Y^c_{t,\mathrm{int}}$, i.e.

$$\hat{X}^c_{t,\mathrm{int}}\Big|_{\mathrm{kg}} = \frac{\hat{X}^c_{t,\mathrm{int}}\Big|_{\mathrm{ha}}}{Y^c_{t,\mathrm{int}}} = \frac{\hat{X}^c_{t,ref}\Big|_{\mathrm{ha}}}{Y^c_{t,ref}} \cdot \frac{Y^c_{o,ref}}{Y^c_{o,\mathrm{int}}} = \hat{X}^c_{t,ref}\Big|_{\mathrm{kg}} \cdot \frac{Y^c_{o,ref}}{Y^c_{o,\mathrm{int}}} . \tag{4}$$

Thus, the generic inputs per kilogram of product solely depend on the yields given for the original country. For a scenario with a yield input in the original country ($Y^c_{o,\mathrm{int}}$) larger (smaller) than the reference scenario ($Y^c_{o,ref}$), generic farming inputs per kilogram $\hat{X}^c_{t,\mathrm{int}}\big|_{\mathrm{kg}}$ turn out to be lower (higher) than the reference value.

The farming inputs per kilogram of wheat are shown in Figures 1c (N–fertilizer input) and d (input from drying). The 5–year averaged wheat yields in the original country (Switzer-

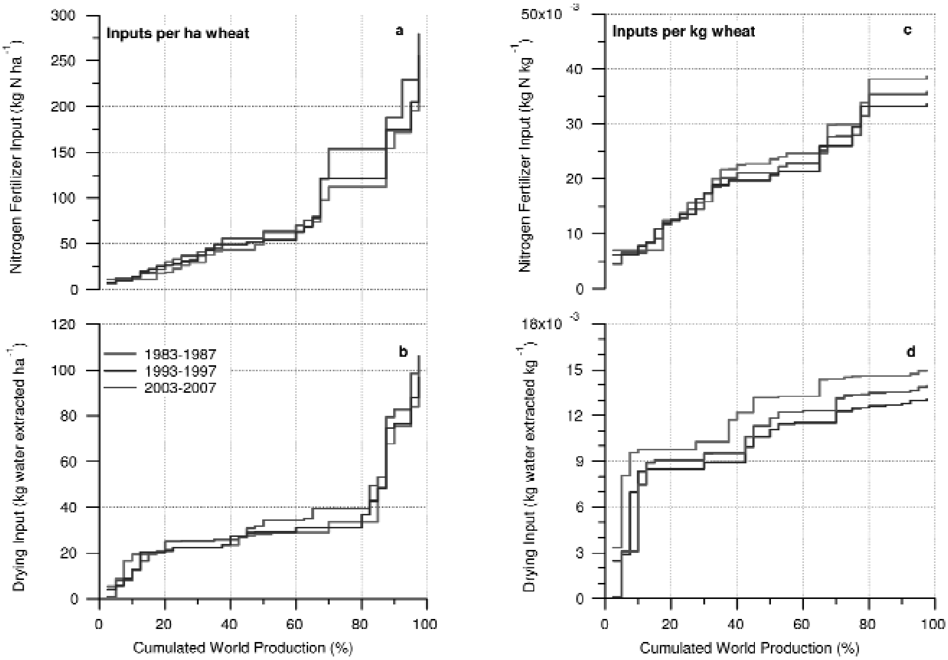

Figure 1: Generic nitrogen fertilizer and drying inputs per hectare (a, b) and per kilogram wheat (c, d) with respect to the cumulated world production as derived with MEXALCA, using Switzerland as the original country. The scenarios assume different yields extracted from FAO (2010) as an input to MEXALCA. Red lines represent results for the reference scenario (yield input averaged over the time period 2003–2007) and blue and green lines for scenarios 1 and 2 where average yields for the time intervals 1983–1987 and 1993–1997 were used, respectively.

land, FAO, 2010) vary between 5350 kg ha^{-1} (scenario 1), 6160 kg ha^{-1} (scenario 2) and 5770 kg ha^{-1} (reference scenario, $Y^c_{o,ref}$). Thus, when applying scenario 1, the factor $Y^c_{o,ref} \big/ Y^c_{o,\mathrm{int}}$ (see equation 4) is larger than 1 and therefore, the extrapolated inputs per kilogram wheat (blue lines in Figures 1c and d) are higher than those derived from the reference input (red lines). Using scenario 2 as an input for the extrapolation, the opposite result is obtained as the yield ratio $Y^c_{o,ref} \big/ Y^c_{o,\mathrm{int}}$ is smaller than 1: The $\hat{X}^c_{i,\mathrm{int}}\Big|_{\mathrm{kg}}$ (green lines) are lower with respect to the reference scenario (red lines in Figures 1c and d).

3.4 Generic environmental impacts per hectare and per kilogram wheat

Applying the same characterization factors for all countries worldwide, each of the generic environmental impacts per hectare can be calculated as the sum of the products of the generic farming inputs per hectare and module and the corresponding impacts per unit of farming input as derived for the original country (Roches *et al.*, 2010). Accordingly, the generic environmental impacts per kilogram of product are obtained by dividing those per hectare by the yield in the target countries.

Figures 2a and b show the generic global warming potential (GWP 100 a, kg CO_2-eq ha^{-1}) and the non–renewable energy demand (MJ-eq ha^{-1}) with respect to the cumulated world

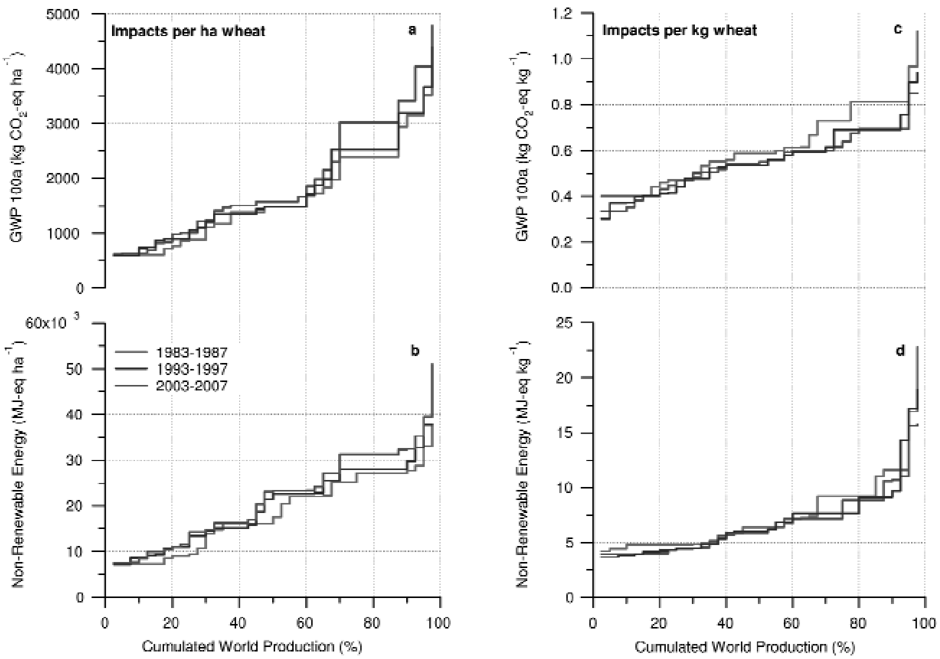

Figure 2: Generic global warming potential (100a) in CO_2–equivalents per hectare (a) and per kilogram of wheat (c) and generic non–renewable energy demand in MJ–equivalents per hectare (b) and per kilogram of wheat (d). Scenarios representing different yield inputs are coloured in the same way as in Figure 1.

production (%). Yield scenario 1 (1983–1987, blue lines) results in the smallest generic inputs per hectare, where weighted averages for both impacts are 19 % lower than for the reference scenario (red lines). Generic impacts per hectare derived from using scenario 2 (1993–1997, green lines) are calculated to be 11 % (GWP) or 9 % (non–renewable energy demand) lower on average than the reference scenario. Both findings are in agreement with the behaviour of the generic inputs per hectare (see Figure 1).

The generic impacts per kilogram of wheat are shown in Figures 2c and d. As described in section 3.2, inputs per kilogram of wheat are largest when driving the model with scenario 1 (Figures 1c and d). Accordingly, the corresponding average impacts per kilogram turn out to be 10 % higher (blue lines) than the reference (red lines) for both GWP and non–renewable energy demand. The generic impacts calculated from driving the model with scenario 2, however, are similar to those obtained from the reference input: weighted averages are only 2 % (GWP) or 4 % (non–renewable energy demand) higher than those of the latter even if the corresponding inputs per kilogram are smaller on average than the reference (Figures 1c and d).

In order to interpret this result it has to be recalled that only some of the key farming inputs (see section 2) are scaled with the yield ratio and others are not. In fact, following scenario 2, generic impacts per kilogram would be 4 % lower for both GWP and non–renewable energy demand if only the yield dependent key farming inputs were taken into account. Furthermore, the generic inputs per kilogram depend on the inverse yield as determined for the original country (Switzerland, see section 3.3) only. Thus, the higher this yield (6160 kg ha^{-1} for scenario 2 in contrast to 5770 kg ha^{-1} for the reference scenario), the lower the contribution of the yield dependent modules to the generic impact per kilogram.

4. Conclusions and outlook

The sensitivity of MEXALCA to variations of crop yields was investigated in a case study for the global production of wheat. The generic impacts per hectare for the categories GWP and non–renewable energy demand were 19 % lower on average than the reference (2003–2007: representing the highest yields) when applying scenario 1 (1983–1987: lowest yields) and 11 % and 9 % for GWP and non–renewable energy demand, respectively, lower when using scenario 2 (1993–1997). This is due to the linear dependency of the inputs and with it, the impacts per hectare on the yield ratio. Driving the model with scenario 1, generic impacts per kilogram are 10 % higher for both impacts (GWP and non–renewable energy demand), while impacts per kilogram of product show minor changes only when scenario 2 is used as an input.

This has two reasons: First, the generic inputs and impacts per kilogram of product are scaled using the inverse of the average yield for the original country, and wheat yields in Switzerland were lowest for the interval 1983–1987 and highest for 1993–1997 (FAO, 2010). Secondly, only some of the estimators applied during the extrapolation of the key farming inputs scale with the yield ratio, while others do not. Thus, the sensitivity of certain generic impacts to the yield also depends on the absolute value of the latter as the effect can be smoothed out by their contribution.

This has to be kept in mind when interpreting the extrapolated impacts: if crop yields in a country rise as a result of increasing farming inputs, this would imply higher environmental impacts, which might be smoothed out and thus not be reflected by the model. Furthermore, the behaviour of crop yields as observed for the original country might not represent the global trend. Thus, the extrapolation would be biased by the conditions of the original country. Accordingly, a next step in the analysis of the sensitivity of MEXALCA to the choice of the data would be the comparison of extrapolation results when using different original countries.

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Estimating emission inventories of French farms at multiple spatial scales using the Farm Accountancy Data Network (FADN)

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ABSTRACT

We used data for the year 2000 from the French Farm Accountancy Data Network (FADN), Agricultural Census, and Annual Agricultural Statistics to estimate energy use and N, P, and K balances of farms in France. FADN data were used to estimate farm-level N-P-K imported in fertilisers and animal feeds, N produced in manure, N-P-K exported in crop and animal products, pesticide input, and energy use. Larger-scale data were used to bound farm-level estimates of fertiliser imports. Based on estimated farm-level inorganic fertiliser input and manure production, emissions of NH_3, N_2O, and NO were estimated using recent emission factors. Emission of CH_4 from enteric fermentation was estimated from expert opinion and applied to FADN data for milk production and livestock units per farm. We present preliminary results of this method and discuss ways to broaden and refine its multi-scale estimates.

Keywords: LCA, Farm Accountancy Data Network, spatial scale

1. Introduction

Generating an emission inventory for the Life Cycle Assessment (LCA) of a single product from an agricultural production system requires a large amount of data. The process becomes more difficult when multiple products, production systems, or spatial scales of production require analysis. As demonstrated by previous researchers (Dalgaard *et al.*, 2006; Thomassen *et al.*, 2009), national and international agricultural databases can provide a majority of data necessary for such analyses, reducing the time spent collecting them. Nonetheless, obtaining farm-level estimates of material and energy flows from these databases can remain difficult, especially when they must be estimated from micro-economic data (e.g., farm-level expenses), regional-scale flow data (e.g., kg of fertilisers purchased in a given state), or both (e.g., fertiliser expenses in a given state). When estimating impacts at larger spatial scales, however, interactions among farms (i.e., one farm's output becomes another farm's input) must be considered.

Dalgaard *et al.* (2006) used data from 2138 farms in the 1999 Danish FADN to create 31 farm types that represented the diversity of soil types (sandy loam vs. sandy), production modes (conventional vs. organic), agricultural products, and livestock densities found in Denmark. After estimating energy use by; N and P inputs to; and N, P, and CH_4 emissions from each of the 31 types, they scaled farm-level emissions estimates up to the national level and noted that the estimates agreed well with national statistics (in most cases, less than a 4% difference). Thomassen *et al.* (2009) used data from 119 farms in the 2005 Dutch FADN to

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estimate environmental impacts of specialised, conventional dairy farms in the categories of land use, energy use, acidification, climate change, and eutrophication. In addition, use of FADN data allowed for calculation of economic-performance indicators for these same farms and calculation of their correlation with environmental impacts. Most impact estimates lay within the ranges of those estimated for dairy farms in other studies, though energy use (MJ per t of milk), as in a previous Dutch study (Thomassen *et al.*, 2008), was significantly higher that that observed in other studies. We attempted a similar exercise in this study, with the added challenge that the French FADN contains fewer data useful for LCA (e.g., quantities of animal feed purchased) than its Danish and Dutch versions.

2. Methods

The majority of data to construct the method came from the year 2000 FADN, which contained a sample of 7758 French farms classified into 68 farm types, which we grouped into 17 classes. Additional statistical data came from the 2000 French Agricultural Census (AC), the 2000 French Annual Agricultural Statistics (AAS), the French Union of Fertiliser Industries (UNIFA), and the French Union of Animal Nutrition Industries (SNIA). For application of the method to other years, data for annual mean prices of inorganic fertiliser and animal feed were included in the database for proper conversion of farm expenses into quantities of material transfer.

2.1. Farm-level inputs

The sources of nitrogen (N), phosphorus (P), and potassium (K) input to farms included in the system boundaries comprised purchased inorganic fertilisers as well as purchased animal feed and forage. To estimate inorganic fertiliser input per farm, N, P, and K concentrations in commercially available fertilisers were obtained (van der Werf et al., 2009; UNIFA, 2009). Then, data at the level of French departments for surface area by crop and number of livestock units (LUs) by type (AAS, 2009) were used as independent variables in a nonlinear regression model to predict N, P, and K in the quantity of fertiliser purchased at the department level (UNIFA, 2009). The mean coefficients obtained from this regression were applied to each farm in the FADN to predict fertiliser N, P, and K inputs from farm-level data for total fertiliser expenses, surface area by crop and forage type, and LUs by type (FADN, 2010). Finally, these farm-level estimates were extrapolated to the national level and then adjusted upward or downward (by the same percentage for all farms) to agree with data for the amount of N, P, and K sold in fertiliser nationwide (UNIFA, 2009).

Similarly, the quantity of N, P, and K imported in animal feed was estimated with a nonlinear regression model predicting the proportion of total farm-level feed expenses spent for each of 12 animal types as a function of farm-level LUs by type, surface area by crop and forage type, crop auto-consumption, forage stocks, and production of milk and eggs (FADN, 2010). Farm-level predictions of expenses per animal type based on this regression then were adjusted upward or downward so that their sum equalled the total feed expenses observed per farm in the FADN. Next, the quantity of N, P, and K in feed purchased for each animal type was estimated by dividing these estimated expenses by the per-tonne price of animal feed and multiplying by the estimated N, P, and K concentrations in the 12 types of animal feed. The quantity of N, P, and K imported in purchased forage, assumed to be hay, was estimated as a linear function of total forage expenses (FADN, 2010) times hay price times N, P, and K concentrations in hay.

Pesticide inputs were estimated from farm-level pesticide expenses (FADN, 2010) multiplied by a mean price per kg of active ingredient. Direct (on-farm) energy use (GJ) was estimated from farm-level electricity, fuel, and natural-gas expenses (FADN, 2010), multiplied by the per-unit prices and energy contents used the analysis tool EDEN-E (van der Werf et al., 2009). Indirect (off-farm) energy use (GJ) was estimated from farm-level expenses for fertiliser, animal feed, electricity, fuel, and natural gas following the procedure used for direct energy use.

2.2. Farm-level outputs

The sources of N, P, and K output from farms included in the system boundaries comprised crop and animal products sold and N emissions in the form of NH_3, N_2O, and NO. Farm-level output of N, P, and K in crop and animal products was calculated for 173 agricultural products based on quantities sold (FADN, 2010) and their estimated N, P, and K concentrations.

The amount of N in manure excreted by livestock was set at 100 kg N/LU/year. Based on estimated farm-level inorganic fertiliser input and manure production, emissions of NH_3, N_2O, and NO were estimated using recent emission factors (Gac *et al.*, 2006), which varied for manure depending upon its location (Table 1). The proportion of cow manure excreted outside of buildings was estimated as the herbivore stocking density of the farm (permanent pasture area (ha) divided by LUs), while sheep, pigs, and goats were assumed to excrete 100, 15, and 15% of their manure outside of buildings. Emission of CH_4 from enteric fermentation was estimated using emissions factors based on expert opinion (P. Faverdin, INRA) applied to farm-level data (FADN, 2010) on milk production and LUs per farm (Table 1). Farm N, P, and K balances were calculated as differences between inputs and outputs of N, P, and K, respectively.

Table 1: Emission factors expressed for N in NH_3, N_2O, and NO as a percentage of farm-level N inputs (Gac *et al.*, 2006) and for CH_4 as a function of milk production and livestock units (LUs)

Source	**N-NH_3**	**N-N_2O**	**N-NO**	**CH_4**
Inorganic fertilisers	3%	1.3%	0.3%	
Manure outside buildings	7%	2.6%	0.2%	
Manure in buildings	15%	0.2%	0.2%	
Manure during storage	9%	0.9%	-	
Manure after spreading	10%	0.9%	-	
Ruminant enteric fermentation				kg CH_4 = 0.92 × L milk + 75.321× LU

The method developed was tested with FADN data from 1992-2007, with results from 2004-2007 reported here.

3. Results and Discussion

Preliminary results of this method provide relatively comprehensive estimates for N, P, and K inputs and N balances (Fig. 1), because the processes influencing them are represented relatively fully. Initial estimates of emissions, such as NH_3, show differences among farm types, especially between specialised crop and livestock farms (Table 2). Nonetheless, estimates of N_2 emissions, biological N fixation, and atmospheric N deposition should be included to improve the accuracy of N-balance predictions. Before attempting to use this approach to predict potential mid-point impacts, the method needs to estimate PO_4 and NO_3

emissions to water for eutrophication impacts, CO_2 emissions (from non-renewable energy, fertiliser, and feed use) and CH_4 emissions from manure for climate-change impacts, SO_2 and NO_x emissions (from energy, fertiliser, and feed use) for acidification impacts, and heavy-metal emissions for terrestrial toxicity impacts. In addition, predictions of CH_4 emission from enteric fermentation should be refined (e.g., IPCC Tier 2).

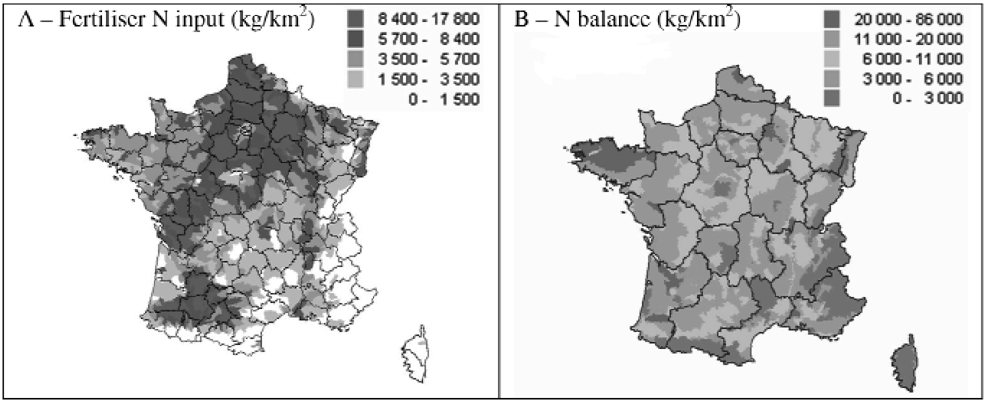

Figure 1: (A) Estimated farm N inputs in inorganic fertilisers by catchment (kg/km^2 total area) and (B) estimated farm N balance by agricultural region (kg/km^2 total area)

Table 2: Estimated total N-NH_3 emissions (kg per ha of usable agricultural area) by farm type from 2004-2007 using French FADN data

Year	Field crops	Vegetables	Horticulture	Vinyards	Fruits, other permanent crops	Dairy and meat cows	Other ruminants	Pigs and poultry	Mixed crops	Mixed crops and livestock	All types
2004	5.9	16.4	50.7	2.9	3.5	23.6	9.6	209.2	11.3	25.8	17.8
2005	6.1	20.4	42.2	3.0	4.0	22.4	10.1	203.1	11.0	25.9	17.5
2006	6.0	19.3	39.1	2.7	3.3	22.2	9.7	180.5	9.0	27.0	17.2
2007	6.2	17.8	33.9	2.8	3.7	22.3	9.3	174.3	9.5	25.0	16.9

One other approach has used the French FADN for environmental analysis: IDERICA, which is based on the analysis method IDEA (Girardin *et al.*, 2004). Because IDERICA predicts qualitative impacts, however, it needed fewer quantitative data than our approach. Certain farm-level data remain difficult to estimate from French FADN data, such as inputs of poultry feed. On vertically integrated poultry farms, farmers do not purchase feed directly; instead, the poultry company supplies the feed and subtracts its cost from the price it pays for the finished poultry.

Once the number of environmental emissions taken into account is increased and existing emission estimates are improved, ways to augment its multi-scale estimates include considering regional transport distances (for fuel use) and the influence of soil and climate on N emissions.

4. Conclusions

The use of French Farm Accountancy Data Network data to estimate farm N, P, and K balances, as well as NH_3, N_2O, NO, and CH_4 emissions, for any year of the survey and at multiple spatial scales (e.g., farm, catchment, department, country) appears a promising approach. It currently requires improvement, however, in the number and precision of nutrient-balance and emissions processes taken into account in the life-cycle inventory stage before it can become an operational tool.

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Resource use and emissions of agricultural production – a top-down approach

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ABSTRACT

This article describes an input/output-analysis of the German agricultural sector that disaggregates the overall emissions and resource uses into production activities and the respective output units, such as one kilogramme of milk. At the national level of the agricultural sector, this analysis is comparable with ´cradle-to-farm gate´ life cycle assessment studies. The methodical and data-technical approach is based on the German environmental-economic accounts, which depict the interdependence of environment and economy. Within this framework of all producing and services sectors, a report module on ´agriculture and environment´ was developed, which offers both a transparent compilation of the monetary values and the physical material flows between agricultural activities and inter-sectoral supplies. A time series analysis from 1991 to 2007 indicates the trends of the resource use and emissions in the German agrarian sector. The presented results show that the agricultural efficiency has increased in the past.

Keywords: Agriculture, Emissions, Input-Output Analysis, Resource Use, SEEA.

1. Introduction

A sustainable use of resources and abatement of emissions into the atmosphere and water bodies are challenges of policy strategies and the legislation at national and European level. In the context of global level climate change, the mitigation of greenhouse gas (GHG) emissions is becoming a priority of environmental policies. For this purpose, GHG sources throughout the production processes have to be better understood. Reliable and broadly accepted calculations are needed to give a strong basis for scientific proofs and political negotiations. Within the debate on climate protection, the accumulated emission values of production activities and of food products are in the centre of the public and political interest.

The purpose of the presented contribution is to introduce an input-output analysis for the description of resource uses and emissions and its application to the German agrarian sector. In contrast to data from life cycle assessments (LCA) for single products, the advantage of this method is the sector-related completeness and consistency to data of the official statistics and environmental reporting. Most important parts of the analysis are the material and energy flows which are connected to the economics as well as to emissions arising from the production processes. The comprehensive description of the agrarian sector offers the basis for the calculation of ecological efficiency indicators based on physical input and output or emission-/output relations.

In this contribution, the analysis instrument for the production, the resource use and the emissions of the German agrarian sector is described. The purpose of this work was to describe the inputs and outputs of the agrarian production in Germany as a whole, and disaggregated by production activities and products. The module on agriculture, coupled with

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monetary dimensions like production values and added value, is integrated into the wider overall economic context of the environmental-economic accounts (EEA).

2. Material and Methods

The EEA, processed by the Federal Statistical Office (Schoer et al., 2000), illustrates the interdependence of economy and environment. It is the German adaptation of the 'System of Integrated Environmental and Economic Accounting ´ (SEEA) which was introduced in 1993 by the United Nations (UN et al., 2003). Its purpose is to deliver environmentally relevant indicators as well as descriptive statistics to the environment and economy. These tools should serve the strategical planning and policy analysis to identify sustainable strategies. The EEA describes German economy by 71 production sectors (PS) whereas the agrarian sector represents one PS. The report module ´agriculture and environment´ of the EEA (Schmidt and Osterburg, 2009) describes the agrarian sector with 30 crop production activities (PA) and 15 PA of animal production. The inputs are calculated by the modelling system RAUMIS (regionalised agricultural and environmental information system for Germany), which integrates official statistics of agricultural structures, inputs, outputs and revenues as well as additional information about resource use and emissions. These data are consistent to the official statistics and international reports.

We intend to generate national average values that can provide reference levels for LCA studies and international comparisons, considering the related system boundaries. The definition of system boundaries is essential for comparison (Ekvall und Weidema, 2004). Some LCA-studies define the life cycle from raw material to farm gate, e.g., production of agricultural commodities before transport and processing (Harris und Narayanaswamy, 2009). The presented module ´agriculture and environment´ of the German SEEA corresponds to these approaches. The second significant influence on the results is the allocation of burdens to the multiple outputs. We use a monetary allocation because of the financial dependency between the production chain and the use of products (s.a. Huppes und Schneider, 1994).

2.1. Data

The advantage of the EEA system is the compilation and arrangement of extensive data from relevant statistics on production activities, input and output quantities, and information on environmental issues. The complexity of material flows and their monetary values are depicted and analysed with the help of models. The report module ´agriculture and environment´ contains two tools: First, specific I/O values of production activities are calculated in the German agricultural sector model RAUMIS for each year and aggregated for the whole sector. Secondly, these results are integrated into an inverted input-output table. The simulation is based on data of the agricultural structures and other statistics of Destatis and the Federal Ministry of Food, Agriculture and Consumer protection (BMELV) as well as on technological and management parameters, derived from farm calculation data or estimated on basis of farm bookkeeping data. Another important data basis are international reports delivering data on gas emissions (UBA, 2009) and nitrogen balances (OECD, 2001). Table 1 lists the most important inputs and data sources as well as outputs of the report module:

A major task was the integration of the various data sources in a consistent system in which all monetary dimensions and physical parameters are brought together. In addition, the data from the agricultural accounts (LGR) and international reports had the uppermost priority. The breakdown of these figures to the level of single production activities had to be adapted for all reported years (1991, 1995, 1999, 2003 and 2007).

Table 1: Input- and Output-Values

resources/inputs:	**data sources**
added value, production values, taxes, subsidies	economic accounts for agriculture
employment of labour	official statistics
energy input	farm data
land use and yields	agricultural statistics
nutrients (N, P, K)	official statistics, OECD database
gas emissions	national emission inventory report (NIR)
Total output amounts, feeding concentrates, imports and exports of agricultural commodities	agricultural statistics
outputs:	
resource use and gas emissions by production activities (e.g. winter wheat, maize, dairy cows, porkers) and important agricultural products (vegetable products, raw milk, meat, eggs)	

2.2. Input-Output-Table

Core element of the input-output analysis is the square matrix which performs all 45 production activities in the first column (delivering processes) as well as in the headline (delivered processes). Figure 1 shows the table structure of the agricultural relations and the peripheral sectors (70 other production sectors (PS) and the final (private) demand). The intra-sectoral exchange of goods between different agricultural activities as well as the inter-sectoral relations between the agricultural sector and other economic sectors can be thereby systematised.

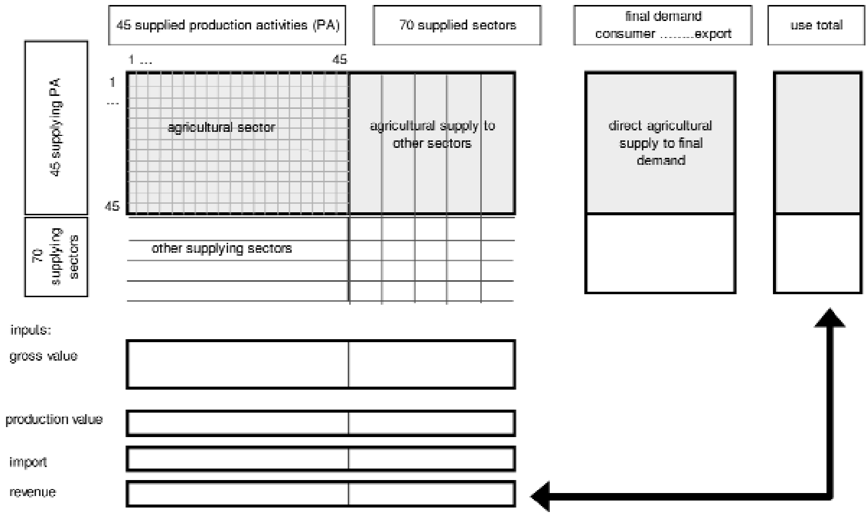

Figure 1: Pattern of the Environmental-Economic Accounting. Source: Schmidt und Osterburg, 2009

The report module of EEA ´agriculture and environment´ is focussed on the production activities of the agricultural sector. This input-output table (IOT) lists all exchanges of intermediate inputs within agriculture and inputs from other sectors, in physical, as well as in monetary unities (ton or euro). The inverted matrix of the IOT, called ´Leontief inverse´, allows the direct and indirect inputs of all producers to be calculated and transfered to output units. The Leontief inverse of the monetary inverse matrix and a subsequent multiplication with a load vector reveals the product specific charges. This results in a load per agricultural product as for example raw milk and meat at farm gate. Load vectors are so-called satellite systems which are complementary to the monetary IOT adding physical data on resources or emissions. The division of the total values with the production amount reveals the specific load per unit (e.g., CO_2/kg). Up to now results exists for the German agricultural sector for the years 1991, 1995, 1999, 2003 and 2007.

3. Results

The inverse matrixes transfer the direct demand of land use and other resources to cumulated results of the agricultural outputs like milk and meat. The example of the land use in 2003 shows this effect in Figure 2. The plant production demand of land area comprises the whole agricultural land (left column) disregarding buildings and holding area. Grassland provides forage through grazing or silage that belongs to the plant products, regarding the direct effect, and to the milk and meat products regarding the cumulated effect (right column). The cumulated value for plant products of about 50 % represents cereals and root crops for processing as well as energy crops.

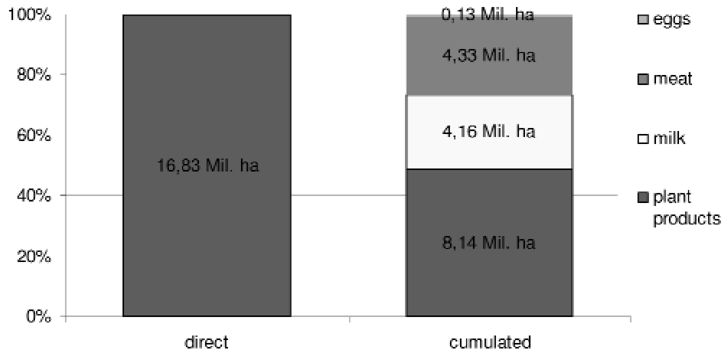

Figure 2: Direct and accumulated surface claims in 2003. Source: Own illustration

The calculations generate physical and monetary totals at the sector level for milk production and other important agricultural commodities. A division through the produced quantities, such as million kilogrammes of milk per year, provides the national average burden per unit of product. The sum of 4.16 Mil. ha of area requirement for milk production and an annual milk production of 28.5 Mil. t results in a specific land use of 1.46 m² for one kilogramme of raw milk. Land for producing imported feed concentrates is not included in Figure 2 but in Figure 3. In the 1990s, the calculated values were at 1.8 – 2 m²/kg, thus milk production has become more efficient in terms of land use. Comparative values from LCA studies are at the same range for conventional farming. Calculations of Swedish conditions (Cederberg and Flysjoe, 2004) range between 1.54-1.92 m²/kg of milk. Results from Netherlands and UK are at a lower level: 1.19 m²/kg (Williams et al., 2006) and 1.3 m²/kg (Thomassen et al., 2008).

Figure 3 shows some results in detail for cereal and milk production. The columns represent the years 1995, 1999 and 2003 and the burden of the agricultural sector (filled out) as well as of the inputs of other sectors and imports (shaded). In the upper diagrams (A, B) direct resource use and emissions of the production units are displayed. The lower diagrams (C, D) show results of the inverse matrix, which generates cumulative direct and indirect effects of the products. In the case of milk production there is no direct land use for dairy cows (comp. diagrams B; area demand of stables is not considered) but for forage cultivation which is displayed in diagram D: area for milk production.

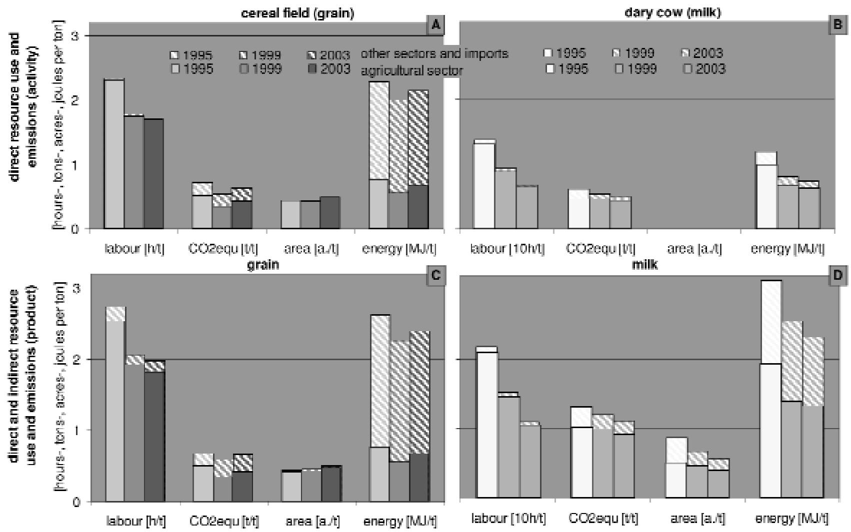

Figure 3: Direct and indirect resource use and emissions of wheat and milk in the German agricultural sector plus delivered products from other sectors and imports. Source: Own illustration

Of course there is a range of uncertainties in the calculation such as the chosen allocation method, but the results are consistent to the official statistical data framework.

4. Discussion and Outlook

The described top-down approach analyses the resource use and the emissions of the German agrarian sector considering economic as well as ecological data which were integrated in a consistent system. The most important data are integrated into a I/O square matrix which takes into consideration all agricultural production areas. A monetary allocation which transfers the load parameters according to the monetary value of the goods was chosen. This approach corresponds to the technical report ISO/TR 14049 that suggests an economic allocation in the case of fixed coupling of outputs (e.g., products of a dairy cow). Also, the most emissions are assigned to the main product with the highest monetary value of the activity (milk production). Indeed, the co-products are also marketed, but to a lower impact on the income (e.g., meat of dairy cow). Co-products of minor value get an equivalent (e.g., organic manure according to mineral fertilizer and its price). If a product is below a monetary limit of 1 % (e.g., hide) it will not be taken into account. One important advantage of this method is the consistent calculation of all relevant data, especially of the statistics and international reporting framework, which delivers a national average value of agricultural commodities. But, this method is not suitable for a more detailed consideration with regard to variations of single production processes or operational considerations. To connect the advantages of a more detailed information with the advantages of the EEA, a combination of IOA and LCA to a so-called Hybrid LCA (Suh and Nakamura, 2007; Weidema et al., 2009) is required. In spite of a preference for monetary allocation in this analysis, a physical allocation might be advantageous when prices or currencies highly fluctuate in space and time.

At this stage of development the agricultural module of the German SEEA we calculate the resource use and emissions of agricultural products at the farm gate and involve inputs of other sectors and imports. Prospectively we intend to expand the system boundaries towards the food and retail industry including processing, storage and transport. The accounting system enables also the integration of private households and the waste management industry to close the production cycle.

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Carbon footprint estimation and data sampling method: a case study of ecologically cultivated rice produced in Japan

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ABSTRACT

Carbon footprint estimation of food products is considered to require collecting data on a number of agricultural producers to ensure statistical representativeness of inventory data. This study evaluated the carbon footprint of ecologically cultivated rice produced in Japan and examined the representativeness of inventory data employing survey sampling theory. Five life cycle stages were set for estimation: raw-material production, rice polishing, distribution and retailing, rice cooking, and waste treatment. Foreground data on over 100 producers were collected in agricultural production. The results show that the carbon footprint of rice is 7.7 kg-CO_2eq/package (4 kg of polished rice). The contribution of raw-material production is considerable, especially that of methane emissions from paddy fields. Representativeness is examined by the standard-error ratio of estimated inputs. The standard error ratio of greenhouse gas (GHG) emissions evaluated by poststratified estimator was 3.8%, which seemed to have enough representativeness. However, the results suggested a smaller sample can improve representativeness if implementing an optimal sample survey.

Keywords: carbon footprint, rice, data sampling

1. Introduction

Japanese activities related to the carbon footprint of products (CFP) started in 2008, and have reached the stage of sale in stores. Regarding carbon footprint estimation of food products, although there still is no consensus on data collection based on statistical theory, researchers may have to survey foreground data on a number of agricultural producers to ensure representativeness of inventory data. This might make CFP in food and agriculture unaffordable, especially for smaller suppliers, or unreliable without reasonable guidelines for data collection on mass suppliers. This study estimated the carbon footprint of ecologically cultivated rice produced in Shiga prefecture, Japan, which is the first product sold in stores to carry a carbon footprint label. In addition, we examined the representativeness of inventory data and the data collection methods, utilizing survey-sampling theory.

2. Estimating the carbon footprint of rice

2.1. Summary of CFP calculation

The product subject to estimation of CFP is specially cultivated polished rice (variety: Koshihikari) produced in the northern area of Shiga Prefecture, Japan (Figure 1). This product is treated with less than one-half the conventional application of chemical nitrogen fertilizer and agrochemicals in rice cultivation. Beginning in January 2010, packages with a CFP

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label have been sold in retailers around Japan. The functional unit in this study is one package (4 kg polished rice). GHG (CO_2, CH_4, and N_2O) emissions were estimated employing a cradle-to-grave analysis.

2.2. System boundaries

Five life cycle stages of rice were set for estimation: raw material production, rice polishing, distribution and retailing, rice cooking, and waste treatment. Figure 2 shows the system boundary of each stage.

Figure 1: Product subject to CFP estimation

In rice polishing stage, both the main product (polished rice) and co-products (rice bran, utilized as fertilizer material) are produced.The environmental loads of both products in the rice-cultivation and rice polishing stages were allocated by economical value.

Environmental load related to durables (agricultural equipment, facilities, cooking equipment, etc.) are not included because of uncertainty about their durable periods. Waste-recycling processes are not estimated in order to avoid double counting with utilization of recycled materials. Transportations of consumers between their homes and retailers are also not taken into account.

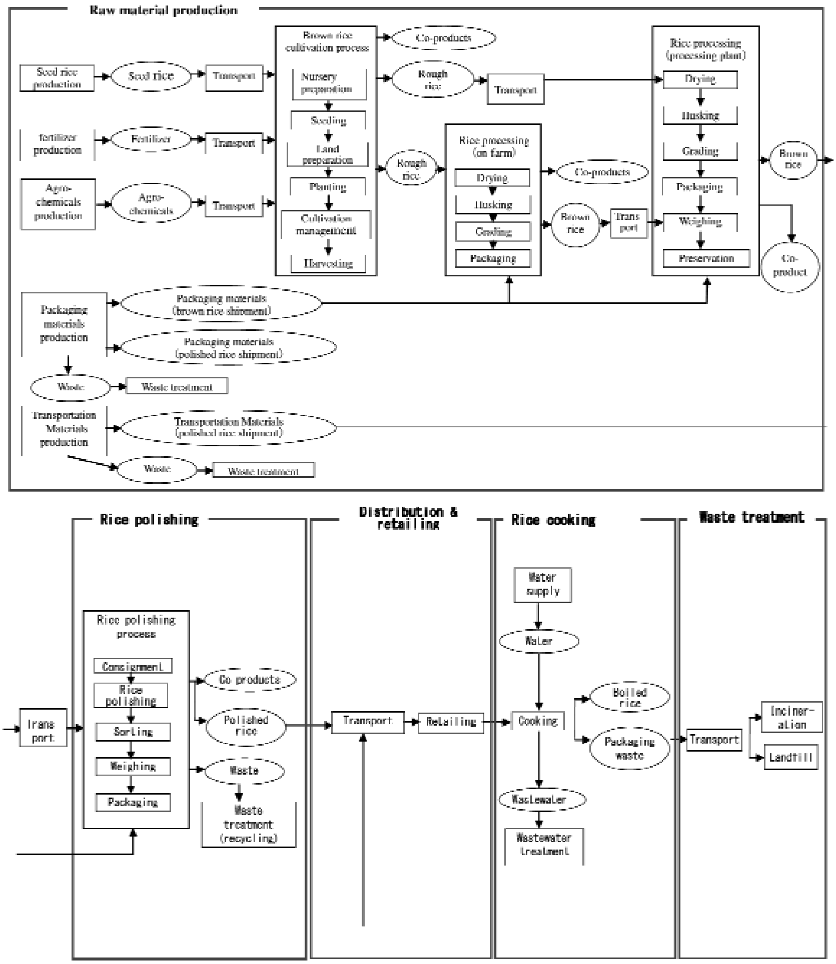

Figure 2: System boundaries

2.3. Data collection

Activity data were collected as foreground data when possible, though some data were collected as background data. Major input materials in each stage are summarized in Table 1.

In the raw material production stage, over 400 producers cultivate the rice for the subject product. This study collected data on 109 producers. These data cover over 50% of all the products, which the current Japanese carbon footprint calculation rules (Product Category Rules, or PCR) for rice require as the standard for data collection. Input data of fertilizer, agrochemicals, fuels, and electricity, in each agricultural producer and rice-processing plant, were surveyed. CH_4 and N_2O emissions from paddy fields also were taken into consideration (GIO, 2009). Actual data of transportation distance were collected for the main product; the distance (500 km) and loading factor scenarios were used for transport of inputs.

Foreground data were surveyed in the rice polishing stage and the distribution and retailing stage. Emissions from the rice polishing stage were calculated from energy usage in rice-polishing plants. Energy use in retailers was collected from chain stores dealing in the subject product. Data on whole stores were allocated to each product by calculating the emission factor per retail price. The average transport distance between stores and rice polishing plants was used for transport of packaged products based on past records of delivery. In the cooking stage, we utilized the PCR scenario, which includes average electricity and water use data in rice cooking using an average domestic rice cooker. In the waste treatment stage, we estimated data for incineration and disposal in landfills of plastic rice packages. The ratio of treatments used is the average value in Japan.

Table 1: Summary of data collection

Life cycle stage	*Inputs*	*Data source of background data*	*Life cycle stage*	*Inputs*	*Data source of background data*
Raw material production	Energy	JEMAI, 2009a	Rice polishing	Energy	JEMAI, 2009a
	Fertilizer	JEMAI, 2009b	Distribution & retailing	Energy	
	Agrochemicals			Transportation	
	Packaging materials	JEMAI, 2009a	Cooking	Energy	
	Seeds	Ajinomoto Co., Inc.,2007		Water supply	
	GHG from paddy field	GIO, 2009	Waste treatment	Waste treatment	

2.4. Results of carbon footprint estimation

Figure 3 shows the results of carbon footprint estimation per package (4 kg polished rice). CFP in all stages is 7.7 kg-CO_2eq/package. About 65% of emissions were related to the raw material production stage; almost all emissions come from agricultural production. CH_4 emission from paddy fields, which is caused by anaerobic fermentation, accounts for 50% of LC-GHGs from agricultural production, although uncertainty concerning its emission factor is high. Besides CH_4 emission, emission of GHGs from fertilizer, energy, and transportation of input materials each accounted for more than 5% of LC-GHGs in agricultural production. After the raw material production stage, the distribution and retailing stage and the rice cooking stage are key stages for emission of LC-GHGs. Most emissions in the cooking stage were from electricity used by rice cookers. All transportation of products and inputs accounted for 6.5% of LC-GHGs.

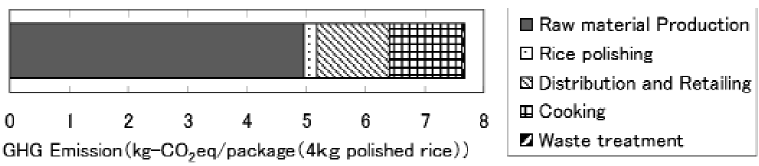

Figure 3: Results of carbon-footprint calculation

3. Evaluation of representativeness of inventory data

3.1. Approach

When calculating the CFP of agricultural production using activity data surveyed by data sampling, uncertainty related to statistical errors in process data becomes an issue, as well as uncertainty of emissions factors and system boundaries. If implementing inadequate data sampling, the cost of surveying CFP rises to ensure reliability of activity data. Sampling survey theory can be applicable for evaluating agricultural activities involving a large number of small producers. This study evaluated the representativeness of CFP data by estimating the variability of calculated data and considered optimal data sampling.

Data variability is examined by the standard error ratio of material input quantity by parent population (all producers), estimated from data on sampled producers. Standard error ratio, corresponding to coefficient of variance of estimates, is evaluated by uncertainty of input data and sampling ratio from parent population. This indicates representativeness of inventory data because both average inventory data estimated from data with high uncertainties and that from few samples have poor reliability to use the data as representative data.

Since cultivated area varies by producer as seen in Table 2, it is assumed that the input quantity of each material correlated with cultivation area. Cultivation area can be more suited for an auxiliary variable than the production, because production changes every year by various factors when cultivation area doesn't change for years. The survey can be designed before harvesting by using cultivation area as an auxiliary variable.

On the other hands, another trend of material input seemed to be found by farm-size level (Table 3). Therefore, this study uses two types of estimation: ratio estimator and poststratified ratio estimator.

Table 2: Distribution and sampling of producers by farm size

	Cultivating area			total
	~2ha	2~5ha	Over 5ha	
Number of producers	81%	14%	5%	100%
Planting area	41%	28%	30%	100%
Sampling ratio (Number of producers)	12%	85%	95%	26%

Table 3: Coefficients of variance in material input of surveyed data

	~2ha	2~5ha	Over 5ha	total
Gasoline	0.97	0.94	0.68	0.95
Diesel oil	0.46	0.91	0.51	0.90
Fertilizer	0.93	0.61	0.63	1.09
Agrichemicals	0.65	0.65	0.90	1.32
N application	0.49	0.34	0.79	1.27

The ratio estimator is the amount of inputs by parent population estimated using inputs by the sample and the ratio between the auxiliary variables of the sample and the parent population. This case utilizes the cultivation area of rice as an auxiliary variable as shown as equation (1).

$$\hat{\tau}_{y_i,R} = \tau_x(\bar{y}_i/\bar{x}) \qquad (1)$$

Where, $\hat{\tau}_{y_i,R}$ is the estimated amount of input y_i, τ_x is the total cultivation area of the parent population, $\bar{y}_i$ is the average input of material i in surveyed producers, $\bar{x}$ is the average rice cultivation area in surveyed producers, and i is the type of input.

Standard error ratio of the ratio estimator is approximated as equation (2).

$$CV(\hat{\tau}_{y_i,R}) = \sqrt{\frac{\tau_x^2}{\bar{x}^2}(1-\frac{n_i}{N})\frac{1}{n_i(n_i-1)}\sum_j (y_{ij}-\hat{R}x_j)^2} \Big/ \hat{\tau}_{y_i,R} \qquad (2)$$

With $CV(\hat{\tau}_{y_i,R})$: standard error ratio of total input quantity estimation of material i, or coefficient of variance of estimates $\hat{\tau}_{y_i,R}$, n_i: number of samples in materials i, N: number of all producers, y_{ij}: total input materials i by producer j, $\hat{R}$: stands for $\bar{y}_i/\bar{x}$, x_j: rice cultivation area of producer j, j: producer..

The poststratified ratio estimator divides the sample into several strata and estimates using a ratio estimator in each stratum. In this case, stratified survey sampling has not been implemented, however, here assumes stratified sampling ex-post facto by utilizing existing sampled data. This study divided the sample into three strata by cultivation area as shown in Table 2. The equation of estimation by the poststratified estimator is shown as equation (3).

$$\hat{\tau}_{y_i,PS} = \sum_d \tau_{x,d}(\bar{y}_{i,d}/\bar{x}_d) \qquad (3)$$

Where, $\hat{\tau}_{y_i,PS}$ is the poststratified estimated amount of input y_i, $\tau_{x,d}$ is the total cultivation area of the parent population in stratum d, $y_{i,d}$ is the average input of material i in surveyed producers of stratum d, and $\bar{x}_d$ is the average rice cultivation area in the surveyed producers of stratum d.

Standard error ratios in poststratified ratio model are also calculated.

In this case, the data representativeness of gasoline, diesel oil, fertilizer, and nitrogen fertilizer application (N_2O), and of agrochemicals, was evaluated because these data were collected by each producer surveyed. The percentages of the sample for which each input datum in the parent population was collected are presented in Table 4. Activity data related to fertilizer and agrochemicals were collected in all surveyed producers because such data on this product are managed by agricultural cooperatives to confirm cultivation standards. However, since energy consumption data have not been collected routinely, the response rate for this data was lower.

The standard error ratio of the total GHG emissions from five material inputs was estimated by Monte Carlo simulation using the standard error ratio of each material as the source of the parameters of the (normal) distribution.

In addition, optimal sampling design was considered in this case. Stratified sampling and Neyman allocation (Optimal alocation) were applied. The number of producers to survey was estimated when the confidence level was 95 %.

Table 4: Sampling ratio by input materials

	Gasoline	Diesel oil	Fertilizer	Agrichemicals	N application
Sampling ratio	16%	16%	26%	26%	26%

3.2. Results of representativeness evaluation

Table 5 indicates the standard error ratio of each material and the total GHG emissions from five materials input. The surveyed data of gasoline is considerably variable, as shown in Table 3, and its uncertainty under both estimation methods is higher than that of other input materials. However, the poststratified estimator performed better than the ratio estimator for other input materials. An especially significant effect of stratification was found in fertilizer and agrochemicals. The standard error ratio of total GHG emissions is about 3.8%, corresponding to a ± 7% confidence interval at the 95% confidence level. This survey is considered to have sufficient reliability in terms of data representativeness.

The required number of samples when implementing stratified sampling is represented in Figure 4. Tolerances are set as 1% and 5% in the 95% confidence level. This is a stricter criterion than the performance that resulted in Table 5. "Total" indicates the minimum sample size that maintains the performance set for all input materials. The number of required samples in the case of 5% tolerance is lower than in the survey actually implemented, although the accuracy is better under optimal sampling that covers smaller producers.

Table 5: Results of standard-error ratio estimations

		Gasoline	Diesel oil	Fertilizer	Agrichemicals	N_2O from N application	*Total*
Standard error ratio	ratio estimator	11.3%	7.6%	11.2%	13.8%	9.1%	5.9%
	poststratified estimator	11.3%	5.8%	5.7%	6.5%	2.3%	3.8%
Average GHG emission ($kgCO_2eq$/10a)	ratio estimator						131.9
	poststratified estimator						138.9

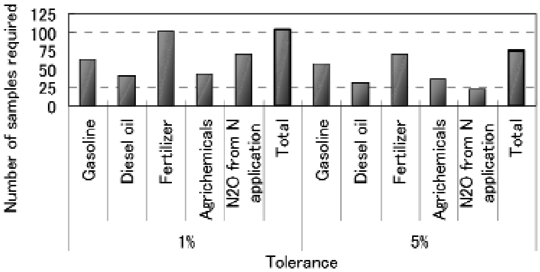

Figure 4: Number of samples required at 95% confidence level

4. Discussion

In CFP calculation, CH_4 emissions made important contributions to total GHG emissions. Although this study could not apply detailed estimation by restriction of data, it is necessary to conduct evaluation in detail including emissions models or measurements, and to make efforts to reduce emissions. The results on transportation of main products and inputs imply the potential effect of local production and consumption, and its limitations.

Although the results of analysis of data representativeness are limited to those consisting of some major input materials, this suggested the importance of implementing sample surveys on CFP for products from a large number of suppliers to improve data reliability and feasibility. This study collected data on variability of material inputs, and these data can be applicable in sample design for CFP of rice produced in situations similar to this case. Also this method can apply to the data quality evaluation in case of data deficiency among large number of producers.

Next step will be including variability of yield in evaluation for precise data quality assessment, because this study evaluated only that of input materials. Besides, further data collection to expand applicability and definition of a framework enabling simple and reliable evaluation of data representativeness will be needed.

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Method for reporting environmental impacts of the Finnish food sector – integration of an IO approach with that of a process based LCA

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ABSTRACT

In Finland the life cycle stages of agriculture represent a major factor in the total environmental load of food chains. The contribution of agriculture in terms of methane, nitrous oxide and ammonia emissions, and nitrogen and phosphorus leaching, is over 90%. The share of agricultural processes is significantly greater than 50% for all the observed classes of environmental impacts. The share of the food processing industry is 0-5% of the chain's entire domestic environmental impacts and the share of other economic areas totals about 6-27%. The contribution of transport on impacts of imported food products is small, representing only 0-6% of the total.

Keywords: carbon footprint of food, national food system, food portion, EIO-LCA approach

1. Introduction

Our research approach was built on an environmental assessment of the food system represented by a national economic input-output model (EIO-LCA), and on process-based LCA models of nutritionally balanced standard lunch plates. The standard lunch plate represents a lunch portion following Finland's current national nutritional recommendations for young people, from 2005. With the lunch plate approach we projected the environmental impacts of food from the micro level, and with the EIO-LCA approach from the macro level. The results of the EIO-LCA approach were integrated with results received from process-based LCAs in order to gain a comprehensive overview on impacts of food consumption in Finland.

The concrete targets of the study were to assess the total impacts attributable to Finnish production and imports of foodstuffs, including their transport, and the specific environmental impacts of standard lunch plates. The ultimate aims were to help consumers make environmentally responsible choices in their future food consumption, to help the food supply chain identify key areas for improvement in terms of various environmental impacts, and to provide policy makers with a tool for monitoring the development of the food sector with respect to use of resources and the potential for climate change impact, acidification, eutrophication and tropospheric ozone formation.

2. Methods

Thirty lunch portions of various compositions were investigated. The impacts of food portion components were assessed through the food chain and environmental impacts, reported phase by phase throughout the production chain. A nutritional serving for a standard lunch

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plate was regarded as a functional unit for calculating the environmental impacts. The lunch plate model incorporates the principle of dividing the plate into three parts; half of the plate comprises vegetables, one quarter protein and the remaining quarter comprises carbohydrate. The plate is completed with a portion of bread and milk. The composition of the dishes took into account the intake of energy (740 kcal), fat (25–35%), protein (10–20%) and carbohydrates (50–60%) in relation to the total energy intake represented by a portion. The serving sizes for the compositions of different food items were adjusted according to the Finnish nutrition recommendations for young people (740 kcal standard lunch plate). Impacts of food raw material production were analysed stage by stage over the whole food chain. In the micro level approach, impacts of different ways of processing food - home, public catering and industrial processes for ready–to-eat dishes - were compared. Impacts of household and post-household activities were also linked to food consumption. Lunch portions are presented in more detail by Saarinen et al. (2010) and food preserving, preparation and home activities for food logistics by Kauppinen et al. (2010).

In the macro level approach the EIO-LCA model used to assess food chain environmental impacts was developed specifically for the Finnish food chain (Virtanen et al., 2009). The model was derived from economic input-output tables of the Finnish national economy associated with environmental emission and characterisation data to compute the environmental impacts. Theoretically the model is a linear economic input-output model, which is expanded with environmental data (see EIOLCA, 2006). The production sectors of the model cover the whole Finnish economy and are, by definition, in accordance with the classification applied to the national account system - except for agriculture, which is disaggregated in order to increase the resolution of hot spot analysis. Imports are modelled product-wise. The classification of the products and the logic for their aggregation to higher level sector-products is convergent to that used in the national account system. The model includes 912 product titles. Disaggregation of the domestic agriculture sector was based on reference inputs and reference environmental loads obtained with the help of a material flow based LCA model built in the study. Sub-sectors were chosen so that each of them has one and only one product as their main product. For each sub-sector of the agriculture-aggregate a share of each aggregate-input and -environmental load was allocated that corresponded to its share of the total of the respective reference flows. Thus the model includes 190 domestic production sectors, of which 44 represent agriculture. The environmental data consist of economic-sector-wise emission data for domestic production, product-wise emission data for imported goods, and emission-specific characterisation data for environmental impact assessment. The model is based on data from 2005. Much of the environmental data was obtained from the material flow and environmental impact assessment model of the Finnish economy, ENVIMAT-model (Seppälä et al., 2009). The emission data for the domestic production sectors are based on national emission inventories and, to a minor extent, on activity information based estimates (Seppälä et al., 2009). For the largest import volumes emission data are life cycle inventory (LCI) data gathered from relevant commercial and public databases and transformed to monetary value specific for the ENVIMAT-model. For the minor import volumes emissions are estimated with the respective domestic emission factors. The model was built by expanding the end-use of the food-chain-products (namely, food products) with their industrial and service usage, and removing the end-use of non-food products from the standard end-use of the national accounts. Sectors that use food products for non-food production, such as the paper industry (starch) and cosmetics industry (sugar), were excluded from the industrial usage. Inter-sectoral feedback was reduced from the gross usage of the industries included in the industrial usage. The rate of feedback was found to be between 0.3 of catering services and 15.8% of vegetables of the gross demand, at an average of 5.8%. Thus, the boundary condition vector of the model was determined solely by the end-use of the food chain products, which consists of the standard end-consumption batch of food products from the national accounts, and of industrial usage, which includes the consumption of the service sector and other sectors of the economy that produce non-food goods as their main products. The end-use of each non-food product was set to zero in the end-use vector. Classification of food chain products follows the definition of foodstuffs given in the EU foodstuff law (Regulation (EC) No 178/2002), according to which foodstuff indicates any substance or product intended to be, or reasonably expected to be, ingested by humans. Additionally, catering services are included in food chain products. The number of food chain products in the end-use vector is 105.

3. Results

According to the results of the IO approach, the food chain accounts for 7% of domestic CO_2 emissions, 43% of CH_4 emissions, and 50% of N_2O emissions, corresponding to 14% of total climate change. The share of the food chain in domestic N-leaching is 58% and that of P-leaching 67%. Of domestic environmental impacts, the food chain is largely responsible for eutrophication of waterways (57%). The contribution of the food chain to climate change was 14%.

In Finland the life cycle stages of agriculture represent a major factor in the total environmental load of food chains. The contribution of agriculture in terms of CH_4, N_2O and NH_3 emissions, and nitrogen and phosphorus leaching, is over 90%. The contribution of agriculture regarding CO_2, NMVOC and NO_x emissions is 30-40% and for SO_2 emissions the proportion is about 23%. PFC compounds are not produced in significant amounts by agriculture. The dominant position of agriculture with regard to these 5 environmental load classes is also reflected in total environmental impacts. The share of agricultural processes is significantly more than 50% for all the observed classes of environmental impacts. The share of the food processing industry is 0-5% of the chain's entire domestic environmental impacts and the share of other economic areas is about 6-27% in total, depending on the impact category.

Total climate change impact per final unit output was found to be highest for domestic meat products, 2.7 kg CO_2 eq/euro and lowest for catering and drink services, 0.6 kg CO_2 eq/euro. The respective value for grain products was 1.8, for vegetable products 1.5, and for fish 1.0. The average for the whole chain was 1.3 kg CO_2 eq/euro, for domestic food products 2.0 kg CO_2 eq/euro (catering and drinks services not included), and for respective imported food products it was 1.8 kg CO_2 eq/euro. For imports the share of animal products (meat and milk products) was 14%, as for the end-use of domestic products it was 44%.

These results were integrated with results received from process-based LCA to develop a comprehensive overview of the impacts of consumption in Finland.

The differences between lunch plates based on animal protein and plant protein were reasonably high; animal protein based lunch plates having at most a five times higher impact on climate change and eutrophication than plates based on plant protein (Fig. 2 and 3).

In addition to the protein component, the impact of the vegetable component turned out to be significant in cases where vegetables were produced in the greenhouse. Finnish greenhouse production currently relies mainly on fossil energy sources. This is an area of challenging technical change, moving towards renewable energy sources and saving energy in Finnish greenhouse production.

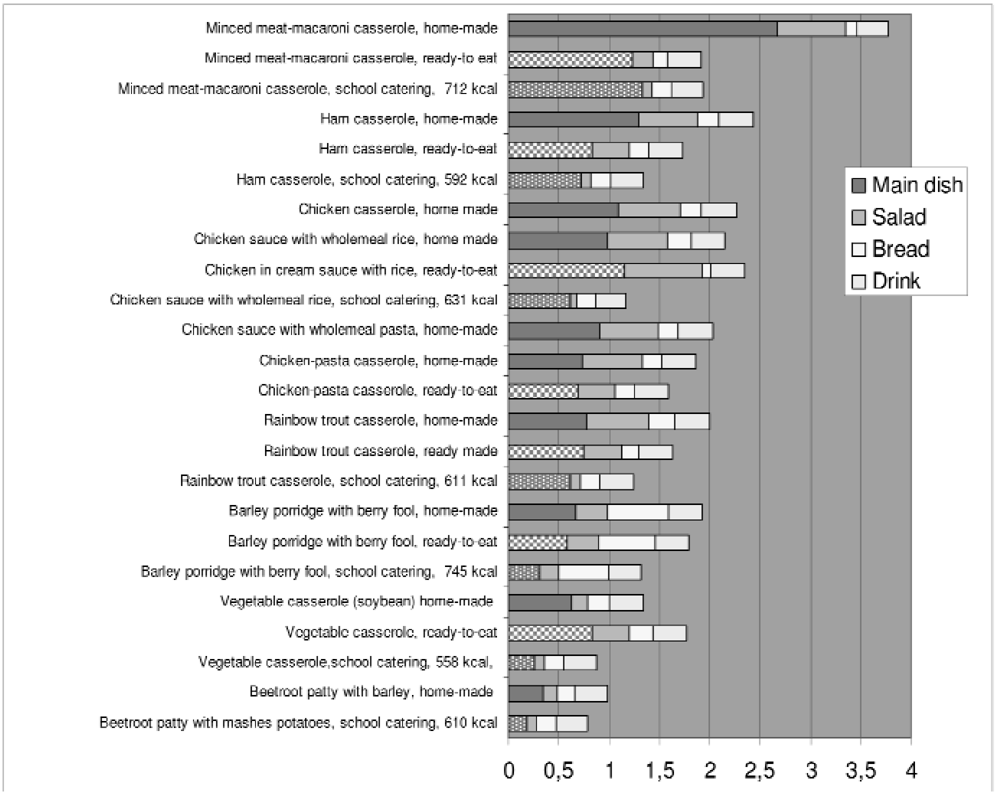

Figure 1. Comparison of climate change impacts of lunch plates in kg CO_2 eq, (home, public catering and industrial processes for ready–to-eat dishes separated with filled, hatched and striped bars). Homemade and ready-to-eat portions 740 kcal, in portions from school catering the energy content defined separately.

The production process had an effect on the environmental impact of the lunch plates, but the effect was not sufficiently large to change the order of the different types of lunch in terms of environmental impacts. We did not establish any major difference between the climate change impacts of industrially processed and home processed food. The main differences originated from differences in recipes for particular lunches when prepared industrially or at home. This observation is in agreement with the fact that the highest proportion of environmental impacts originates from production of raw material. A general rule of thumb can thus be drawn from the different types of lunches to help steer food choices towards broad scale reduction of the potential impact on the environmental load.

However, when the impact of the food chain on climate change is reasonably small, the total impact of diet change does not significantly alter total national scale impacts, as suggested by Risku-Norja et al. (2009). The contribution of diet to eutrophication impact is much higher, as average eutrophication impact of a Finn is 9.4 g PO_4 per day (Nissinen et al., 2006) Thus one lunch represents about 10 to 30% of the daily eutrophication impact of a Finn. As lunch is supposed to constitute one third of daily nutrition, an average contribution of daily food intake represents 30 - 90% of daily eutrophication impact, which agrees well with the value of 57% produced from the IO approach.

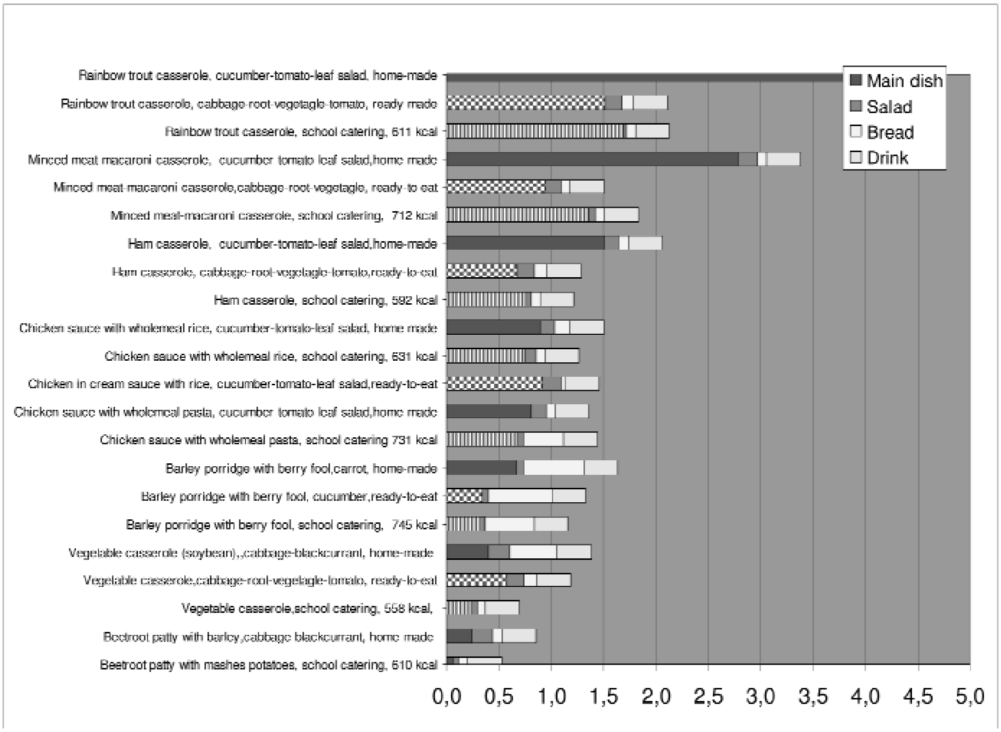

Figure 2. Comparison of water eutrophication impacts of lunch plates, in g PO_4- eq (home, public catering and industrial processes for ready–to-eat dishes separated with filled, hatched and striped bars). Home made and ready-to eat portions 740 kcal, in portions from school catering the energy content defined separately.

4. Discussion

There was fairly good agreement between the values from the micro and macro level approaches. The applicability of the results from the two approaches differs in some respects however. The IO based approach relates the impacts to society as a whole and furnishes an idea of the magnitude of changes that can be realistically realised. In some cases it might be prudent to continue with fairly high-impact sub-sectors if the impacts related to other potential production areas are lower. Another reason for continuation might include situations when there is an excess of resources, such as water for instance, the efficient utilisation of which supports a decision for continued production.

The lunch plate approach focuses on consumption, which is associated with aspects of nutrition and international equity. In terms of equity, excessive resources should not represent an excuse to ignore the requirements for pursuing an environmentally friendly diet. Thus in certain cases linkages between impact of national consumption and national production weaken. In such cases, and to an increasing degree in our globalised food system, there should be a broader focus, and when assessing the relationship between food consumption and production, national borders should not contain spatial production and food systems. Another option would be to focus more intensively on local raw materials for food, but changes made in any particular area would have to be fairly comprehensive in order for them to have a measurable impact.

The complete reports on this work were published in Saarinen et al. (2010) and Virtanen et al. (2009).

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