The environmental impact of pork production from a life cycle perspective

Ph.D. Thesis
by
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Preface and acknowledgements

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Foulum, September 2007

Randi Lundshøj Dalgaard

CONTENTS

Papers included

The following five papers are included in the thesis:

Paper 1. Randi Dalgaard, Niels Halberg, Ib Sillebak Kristensen & Inger Larsen. 2006. Modelling representative and coherent Danish farm types based on farm accountancy data for use in environmental assessments. *Agriculture, Ecosystems & Environment. 117(4), 223-237.*

Paper 2. Randi Dalgaard, Jannick Schmidt, Niels Halberg, Per Christensen, Mikkel Thrane & Walter A. Pengue. 2007. LCA of Soybean Meal. *International Journal of Life Cycle Assessment. (In press).*

Paper 3. Randi Dalgaard, Jørgen Dejgaard Jensen, Bo Weidema, Niels Halberg and Claus Å.G. Sørensen. Environmental assessment of Danish pork. *Submitted to International Journal of Life Cycle Assessment.*

Paper 4. Per H. Nielsen, Randi Dalgaard, Arne Korsbak & Dan Petterson. 2007. Environmental assessment of digestibility improvement factors applied in animal production – a case study of Ronozyme WX CT xylanase applied in Danish grower pig production. I*nternational Journal of Life Cycle Assessment. (In press)***.**

Paper 5. Randi Dalgaard. Slurry technologies and their potential for environmental improvements. *(Manuscript in preparation).*

Summary

The global pork production is forecast to increase significantly over the coming decades and this will obviously affect the environment. The overall aim of this thesis was to improve the understanding of the environmental impact of Danish pork production in a global context and to suggest improvements at the most important environmental hotspots.

Life cycle assessment (LCA) was used as an environmental assessment tool, and the consequential modelling approach was applied. The pig farm was identified as being the most important environmental hotspot in the product chain of pork in relation to both global warming, eutrophication and acidification. The production of artificial fertiliser and pig feed outside the pig farm was also an important contributor to global warming, whereas the emissions from the slaughterhouse and transport from farmer to retailers were low. The largest contribution to global warming came from nitrous oxide, primarily emitted from fertiliser and from denitrification of nitrate. The largest contributions to eutrophication and acidification came from nitrate and ammonia respectively. All these substances contain nitrogen. Thus a more efficient use of nitrogen at the pig farms and in pig feed production will improve the environmental profile of pork.

It was concluded that the global warming potential per kg pork could be reduced by approximately 5% if the digestibility-improving enzyme xylanase was added to the pig feed, whereas the reduction in eutrophication potential per kg pork was limited. The reduction in greenhouse gases was primarily due to the saved pig feed.

A farm model was developed in order to explore to what extent the environmental impact could be reduced if the slurry was separated into a liquid and fibrous fraction or if the slurry was anaerobically digested and the biogas was used for heat and power production. It was concluded that if a pig farm was situated in a region with high livestock density and was obliged by legislation to export slurry to another farm, the amount of P applied to the fields at the pig farm and the amount of slurry transported could be reduced by 85 and 37% respectively, if the slurry was separated and the fibrous fraction exported from the pig farm. However these environmental improvements required that the slurry separation plants were separating efficiently, and this was not the case at the two private farms where data were collected.

Although slurry separation resulted in less transport and use of artificial P at the slurry-receiving farm, the reduction in greenhouse gas emissions was very limited compared to the greenhouse gas emissions from the remaining parts of the product chain of pork. In contrast to this, it was estimated that the global warming potential per kg pork could be reduced substantially if the slurry was anaerobically digested. On the other hand, anaerobic digestion of slurry did not have the same potential as slurry separation had for reducing the P application to fields of the pig farm.

Finally, it was concluded that there is a need to develop the methodologies for quantification of nitrous oxide, phosphate and land-use change related CO_2 emissions in order to further improve the quality of LCAs on agricultural products.

Summary (Danish)

Verdens svinekødsproduktion vil stige markant i løbet af de kommende årtier og dette vil påvirke miljøet. Formålet med denne afhandling var at øge indsigten i den danske svineproduktions miljøpåvirkning i en global sammenhæng, samt at foreslå forbedringer i de dele af produktionen, som er mest miljøbelastende.

Livscyklusvurdering (LCA) blev anvendt som miljøvurderingsværktøj. Svinebedriften viste sig at være den mest miljøbelastende del af svinekødets produktkæde, set i forhold til både global opvarmning, eutrofiering og forsuring. Kunstgødnings- og foderproduktionen bidrog også betydeligt til den globale opvarmning, hvorimod bidrag fra slagteri og transport af svinekødet var lavt. Det største bidrag til global opvarmning kom fra lattergas, som primært blev udledt fra gødning og ved denitrifikation af nitrat. Det største bidrag til eutrofiering og forsuring kom fra henholdsvis nitrat og ammoniak. Alle disse forbindelser indeholder kvælstof, og en oplagt måde at forbedre svinekødets miljøprofil på er derfor at effektivisere kvælstofforbruget på svinebedrifterne og i produktionen af svinefoder.

Det blev konkluderet, at det globale opvarmningspotentiale per kg svinekød kunne reduceres med ca. 5%, hvis det fordøjelighedsfremmende enzym xylanase blev tilsat svinefoderet, hvorimod xylanases effekt på eutrofieringspotentialet var begrænset. Reduktionen i drivhusgasudledning skyldtes primært det lavere foderforbrug.

En gårdmodel blev udviklet for at analysere i hvor høj grad miljøpåvirkningen kunne reduceres, hvis svinegyllen blev separeret i en flydende fraktion og en fiberfraktion, eller forgasset i et biogasanlæg, hvor biogassen efterfølgende blev anvendt til varme og elektricitetsproduktion. Det blev konkluderet, at på en svinebedrift i et husdyrtæt område, som pga. lovgivning skal transportere gylle til andre bedrifter, kunne mængden af P tilført markerne på svinebedriften og mængden af transporteret gylle reduceres med henholdsvis 85 og 37%, hvis gyllen blev separeret og fiberfraktionen eksporteret ud af svinebedriften. Disse miljøforbedringer kræver dog at gylleseparationseffektiviteten er høj, og dette var ikke tilfældet på de to private bedrifter, hvorfra data var indsamlet.

Selvom gylleseparationen medførte mindre gylletransport og lavere forbrug af fosforkunstgødning på den modtagende bedrift, var reduktionen i drivhusgasudledningen meget begrænset set i forhold til mængden af drivhusgasser udledt fra de øvrige led i svinekødets produktkæde. Bioforgasning af gyllen med efterfølgende elektricitets- og varmproduktion viste sig derimod at kunne reducere drivhusgasudledningen per kg svin betragteligt. Til gengæld havde bioforgasning ikke det samme potentiale for at reducere P-tilførslen til markerne på svinebedriften, som gylleseparation havde.

Afslutningsvist blev det konkluderet, at der er brug for videreudvikling af metoder til at kvantificere udledningerne af lattergas og fosfat, samt CO_2 udledninger forårsaget af ændret arealanvendelse for yderligere at forbedre kvaliteten af fremtidige livscyklusvurderinger af landbrugsprodukter.

1 Background and aim

When a pork chop reaches the refrigerated counter in the supermarket it has accomplished a long journey. First sows are raised to produce piglets, feed for the pigs is grown, harvested and transported. Next the pigs are fed, slurry is excreted and then applied to the fields. The pigs are transported to the slaughterhouse, slaughtered, carved up and finally the pork chop is brought to the supermarket, from where it ends up in the shopping basket of a consumer and finally on a dinner plate. In each of these steps energy is used and pollutants are emitted. For example, artificial fertiliser is applied to the field where pig feed is grown and energy is used to produce this artificial fertiliser. In addition, different pollutants, e.g., nitrate and nitrous oxide, are emitted when the pig feed is grown or when slurry is excreted from the pig. Transport of fertiliser, pigs and feed results in emission of CO_2 and other substances. All in all, many different kinds of pollutants in different amounts are emitted before the pork chop is ready for consumption. These pollutants contribute to climate change, eutrophication (nutrient enrichment), increasing acidity in the aquatic environment, changes in biodiversity or other undesired impacts on the environment.

The world population is forecast to grow by 37% from 2006 to 2050 (United Nations, 2007), and this growth obviously requires an increase in food production. The production of meat is forecast to reach 465 million tonnes in 2050, which is more than double the amount of meat produced in 1999/01 (Steinfeld et al., 2006). The growth rate of the world's population is lower compared to the growth rate of the livestock sector, and this reflects that the meat consumed per capita will increase. However, the production of monogastric animals (pigs, poultry), which are mostly produced in industrial units is forecast to grow more rapidly than the production of ruminants (cattle, sheep, goats), which are often raised extensively (Steinfeld et al., 2006). At the global scale more pork is produced than poultry, and pork is also forecast to top the world's meat production in 2015 (FAPRI, 2006). To meet the increased demand for pork, more pig feed will be produced, more deforestation will occur (Steinfeld et al., 2006), more slurry will be excreted and more pig meat will be transported. Consequently, a cascade of polluting activities will be stimulated by the increased demand for pork.

Can the pig sector and the food industry meet this increased demand without increasing the environmental impact at an equal rate? The answer cannot be given with our current knowledge. But a step towards an answer is to identify the most polluting areas of the activities, the so-called environmental hotspots of the pork's product chain and to estimate the potential of improvements within the product chain of pork.

1.1 Structural changes in the Danish pig sector

Denmark has been the world's largest exporter of pork since 1993 (FAO, 2007), and produces 25 million pigs yearly (Danish Meat Association, 2007). Denmark has one of the highest pig densities in the world (Steinfeld et al., 2006), and Danish farmers produced 598 pigs per km^2 and 4.7 pigs per capita in 2006 (Statistics Denmark, 2007). The Danish pig sector has undergone large structural changes over the last decades (Kristensen & Hermansen, 2002), as has been seen in many other countries (OECD, 2003). Figure 1 shows the development in the number and size of pig farms for the period 1996-2006. The number of pig farms has fallen by 40% (from 3980 to 2402), while the average farm size has more than doubled during the last ten years. In the same period the Danish pig herd has increased by 23% (Danish Meat Association, 2007).

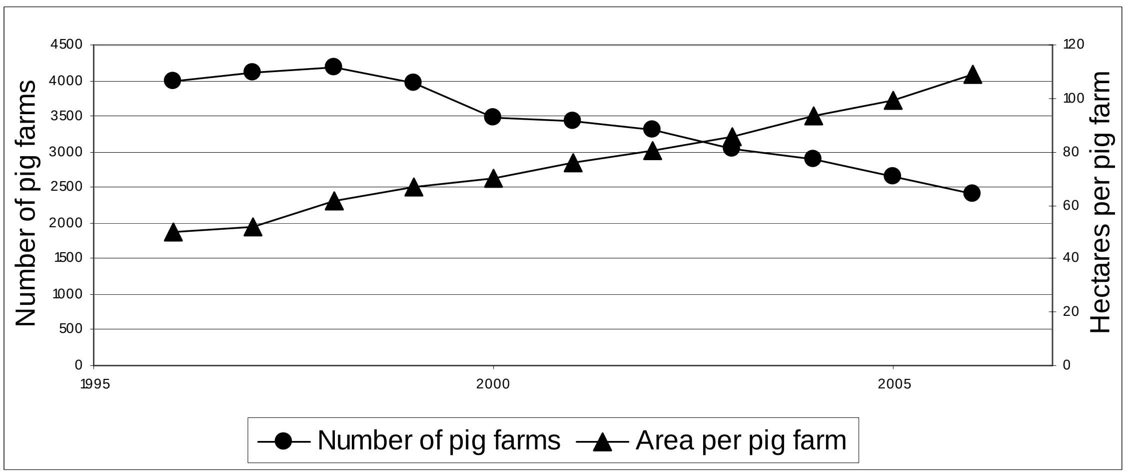

Figure 1. The development in number and size of pig farms in Denmark from 1996 to 2006. (Data is from Danish Statistics (2007)).

The pig farms are also becoming more specialised. Twenty years ago the majority of the pig farms had both sows and fattening pigs, while now the majority has exclusively fattening pigs and no sows, and only a minority has a mix of sows and fattening pigs or only sows (Statistics Denmark, 2007). Moreover, the pig farms are increasingly concentrated in specific geographical areas (Kristensen & Hermansen, 2002). Figure 2 shows how the pigs are concentrated in specific regions of Denmark. In the eastern part of Denmark (the island Zealand) only very few areas have more than 0.8 pig livestock units per hectare, whereas in Jutland (large peninsula in the western part of Denmark), there are several large areas with more than 0.8 pig livestock units per hectare. Figure 3 shows the distribution of total livestock units per hectare (incl. poultry, cattle and mink) and again Jutland has a higher livestock density compared to the islands. This centralisation of the livestock in specific regions may put pressure on the local environment, because ammonia is evaporated from the slurry and the slurry is applied as fertiliser to the fields.

The overall tendency is clear: Compared to previous years, Danish pigs are now produced at fewer, larger and more specialized farms that are centralised in specific regions. In that sense the pig sector has many things in common with industry, and this might explain why the term ‘pig factory’ is often used in the public debate. In the following it will briefly be explained in what way the Danish pig sector affects the environment.

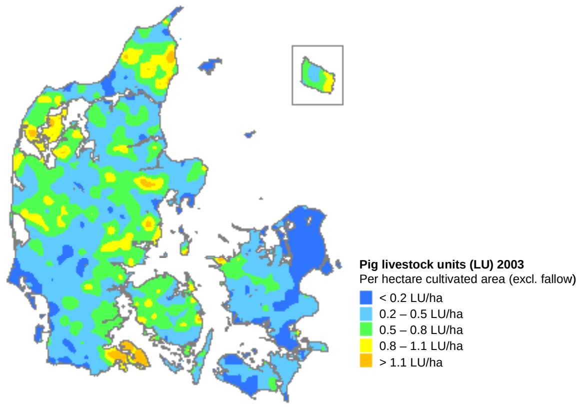

Figure 2. Pig livestock units per hectare in Denmark in 2003 (Djf Geodata, 2007). One livestock unit equals a yearly production of 36 fattening pigs (size: 30-100 kg).

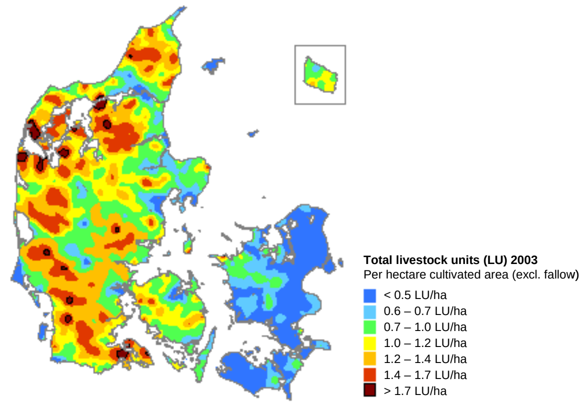

Figure 3. Total livestock units per hectare in Denmark in 2003 (Djf Geodata, 2007). One livestock unit equals a yearly production of 36 fattening pigs (size: 30-100 kg).

1.2 Pig production and the environment

Several kinds of pollutants that negatively affect the aquatic and terrestrial environment are emitted from pig farms, and many of these can be divided into N and P compounds. The N compounds include ammonia, which evaporates from the slurry in the pig house, when the slurry is stored, and after it is applied to the field. A typical Danish pig farm emits 27-44 kg ammonia-N per hectare per year (Paper 1), most of it from the pig house. The ammonia can be deposited in vulnerable zones where it might decrease species richness because of eutrophication. Ammonia also has an acidifying effect and can affect natural habitats, some of which may be transboundary (e.g., lakes in Sweden). Nitrate is another important N compound and 63-95 kg nitrate-N per hectare per year (Paper 1) is typically leached from the fields at Danish pig farms. Nitrate can be leached to the surface water or the ground water, thus it can cause both nutrient enrichment of the aquatic environment or pollution of drinking water.

The only P compound from the pig farms, which has a direct effect on the environment, is phosphate which can be leached from the fields or transported by erosion with soil particles. Most of the P that is applied to the fields is sorbed to soil particles (Poulsen & Rubæk, 2005) and 1.2-2.2 kg P per hectare per year is leached from typical pig farms (Paper 1). In general, more phosphate is leached from livestock farms than from cash crop farms. P is the limiting factor for algal bloom in most of the Danish lakes (Kronvang et al. 2001), and is therefore unwanted in excess in the aquatic environment. Pesticides and their residues also affect the aquatic and terrestrial environment, but due to differences in their toxicity and degradability, some have a larger environmental impact than others.

Similar to the aquatic and terrestrial environment described above, the atmospheric environment is also affected by pollutants emitted from pig production. From agricultural production, nitrous oxide, methane and CO_2 are the most important contributors to global warming (Olesen, 2005; Paper 1). Nitrous oxide is emitted from slurry handling and from fields. On an annual basis, 4.5-5.1 kg nitrous oxide-N per hectare is emitted from a typical Danish pig farm (Paper 1), and although this is a small amount compared to ammonia and nitrate emissions, the contribution to global warming is significant, because nitrous oxide is a very strong greenhouse gas. Methane is also emitted from slurry handling and storage, while fossil CO_2 is emitted from the combustion of fossil fuels. Finally, CO_2 can be emitted from the soil if more organic matter is degraded than applied to the soil.

In addition to the above described environmental effects of pig production, there are also some indirect effects. For example, Danish pigs consume large amounts of soybean meal from soybeans cultivated in South America, resulting in deforestation, greenhouse gas emissions and loss of biodiversity (Paper 2; Tengnäs & Nilsson, 2003; Steinfeld et al., 2006), but these environmental impacts are not targeted by the environmental regulations in their present form.

The so-called ‘Environmental Technologies’ have gained ground in the Danish pig sector during the last 10-15 years. Examples of these ‘Environmental Technologies’ are: i) slurry separation that facilitates the transport of slurry out of areas with a high livestock density (Paper 5); ii) anaerobic digestion of slurry which reduces the greenhouse gas losses from the slurry when it is applied to the fields (and also substitutes fossil energy (Paper 5)); iii) acidification of the slurry which decreases the ammonia loss; and iv) addition of digestibility-improving enzymes to reduce the feed consumption and slurry excretion (Paper 4; Nielsen & Wenzel, 2006). However, it is not always clear to what extent these ‘Environmental Technologies’ can reduce the pollution caused by pig production.

The pig farms’ impact on the environment is often the subject of media attention, and so is the environmental regulations’ impact on the pig farmers. The most important aspects of the environmental regulation of the Danish pig sector will be described in the following.

1.3 Environmental regulation

The concern about nutrient losses from the agricultural sector in Denmark increased in the 1970s and 1980s. This concern hit the political agenda and in 1986 the ‘NPO Action Plan’ was launched. It aimed at reducing N- and P-pollution and was followed by three Action

Plans for the Aquatic Environment (1987, 1998 and 2004), the ‘Action Plan for a Sustainable Agriculture’ (1991) and the ‘Ammonia Action Plan’ (2001) (Dalgaard et al., 2004).

These action plans dictate several restrictions. For example, farmers are not allowed to apply slurry to the fields during winter and must therefore have large storage capacity (Anonymous, 2006). With the purpose of reducing the ammonia emissions, the regulations stipulated, among other things, that slurry tanks should be covered, that slurry must not be broadcast above ground and must be incorporated into the soil within 6 hours after application, etc. (Anonymous, 2006). The environmental regulations focused specifically on the handling and use of slurry, justified by the fact that more than 50% of the nutrients used for crops in the Danish agricultural sector are in the slurry (Petersen, 2007).

Numerous other restrictions exist, but in the following only the ‘Nutrient farm account’ system and the requirements for the farmers to keep a balance between the number of animals on his/her farm and the size of the area under cultivation will be explained.

The legislation (Anonymous, 2006) limits the amount of slurry that can be applied per hectare. On a pig farm an amount of slurry corresponding to 1.4 livestock units can be applied per hectare per year and for a cattle farm this is 1.7 livestock units per hectare. All farmers are obliged to forward a ‘Nutrient farm account’ to the authorities every year, with information on the quantity of slurry-N produced, exported and imported and amount of artificial fertiliser-N imported to the farm. Other agricultural information such as land area, crop types, crop rotations must also be reported. The intention of the nutrient farm account system is to limit the use of nitrogen (N) applied to the crops and thereby reduce the N losses to the environment. Therefore each farm has a fertiliser N-quota, which is based on an N norm for each crop type and adjusted by soil type. For example, in 2007/2008 a cash crop farmer on a sandy loam soil with no slurry production could import 151 kg N in artificial fertiliser per hectare of winter barley (Plantedirektoratet, 2007). Slurry-N also counts in the N-quota, hence a livestock farm must purchase less artificial fertiliser-N compared to a cash crop farm (papers 1 and 5). A pig farm with more than 1.4 livestock units per hectare (equals 47 fattening pigs from 30 to 100 kg produced per year) is obliged to export part of the slurry from the pig farm, and report the name of the receiver of the slurry to the authorities. The receiving farm also has an N-quota and must as a consequence of the slurry import buy commensurately less artificial fertiliser-N. This ensures that slurry-N application is limited, although the slurry is exported from the pig farm.

As previously explained, the average size of Danish pig farms increases over time (figure 1). If a pig farmer with a high livestock density wishes to expand the pig herd, he/she is obliged to either buy more land or to arrange slurry agreements with farmers who can receive the extra slurry. However, this might be expensive, especially if the expanding farm is situated in an area with a high livestock density, where there is competition for land and slurry agreements with farmers that can receive the slurry. As shown in figure 3, large areas have livestock densities above 1.4 per hectare, and therefore the slurry has to be transported long distances. An increasing number of farmers are investing in slurry separation technologies (Birkmose, 2007), because it facilitates the export of slurry from the farm. When slurry is separated it is split into fractions with different nutrient contents. For example, some of the slurry separation plants can produce a fibrous fraction that contains 12 kg N per tonne (Birkmose, 2004), which is higher than the 5-6 kg N per tonne in raw slurry (Anonymous, 2007a). Thus, the amount of slurry transported can be halved if the farmer invests in a slurry separation plant. For a pig farmer who already has more than 1.4 livestock units per hectare, investment in a slurry separation plant can be an alternative to investment in more agricultural land.

It should be emphasized that the 'Nutrient farm account' system, which controls the exchange of slurry between farms and aims at reducing the N losses from the agricultural sector, only focuses on N. Although slurry also contains P, and phosphate is a threat to the aquatic environment (Poulsen & Rubæk, 2005; Kronvang et al., 2001), there are no restrictions on the amount of P (slurry or artificial fertiliser) applied to the fields. However, in the Third Action Plan for the Aquatic Environment, there are some attempts to reduce phosphate leaching (e.g., tax on P mineral feed).

Greenhouse gas emissions is a hot topic in the global debate, and although the contribution from agriculture to the Danish greenhouse gas emissions inventory has been estimated at 18% (Olesen, 2005) there is no national action plan that aims directly at mitigating greenhouse gas emissions from the agricultural sector. The environmental legislation and regulations in Denmark also exclusively focus on farms, although it is well-documented that – for example – the yearly consumption of approximately 1,650,000 t soybean meal in Denmark (Danish Statistics, 2007) contributes to global warming and loss of biodiversity (Steinfeld et al., 2006; Paper 3). At the international level the Kyoto Protocol was negotiated in 1997, and according to this more than 160 industrialized nations have committed to reducing their greenhouse gas emissions by 2012 (Anonymous, 2007b). Each year each country yearly estimates and submits a National Greenhouse Gas Inventory to the UN Framework Convention on Climate Change (UNFCCC) and EU (e.g., Illerup et al., 2007; Anonymous, 2005).

To assess to what extent an agricultural production affects the environment is not simple, and one of the first tasks is to choose a proper environmental assessment tool that can improve the knowledge on environmental impact of current production system and find solutions to reduce these impacts. Some of the most relevant tools will be described in the following section.

1.4 Environmental assessment tools

Different types of assessment tools have been developed to establish environmental indicators, which can be used to determine the environmental impact of livestock production systems or agricultural products. The environmental assessment tools can be divided into the area-based or product-based as argued by Halberg et al. (2005). Area-based indicators are for example 'nitrate leached per hectare' from a pig farm (e.g., Paper 1), and product-based indicators are for example 'global warming potential per kg pork' (e.g., Paper 3).

As explained in Paper 1, the area-based indicators are useful for evaluating farm emissions of nutrients such as nitrate, ammonia and phosphate that all have an effect on the local environment, and area-based indicators have – for example – been used to compare nutrient surpluses from different farm types (e.g., Kristensen et al., 2005; Watson et al., 2002). In a situation where a farm is situated in a nitrate vulnerable zone it is obviously useful to assess the amount of nitrate leached per hectare from the farm to clarify to what extent the farm impacts the local environment, and in that case the area-based indicators are useful. On the other hand, when considering the greenhouse gas emissions from the agricultural production the area-based environmental assessment tools must be used with caution, because global warming is a global and not a local effect. Greenhouse gases impact the climate, irrespective of whether they are emitted from a Danish farm or from a soybean field in Argentina. An exaggerated example could be the dairy farmer who wants to reduce the emissions of fossil CO_2 from tractor driving, and therefore in the efforts to save diesel gets lower silage yields, and therefore has to import more fodder to the farm. If this imported fodder has a higher greenhouse gas emission per tonne produced than that produced on the dairy farm, the savings of diesel for the tractor is a bad solution. In this case, the area-based environmental indicator would not have revealed that the diesel savings could not reduce the greenhouse gas emissions, but the product-based environmental indicator would. Thus product-based indicators are useful for evaluating the impact of food productions on the global environment (e.g. climate change) and have the advantage that in addition to emissions from the farms, emissions related to the production of inputs (e.g. soybean, artificial fertiliser) and outputs (e.g. slurry exported to other farms) are also included. In that way it is easier to avoid 'pollution swapping', which means that the solving of one pollution problem creates a new.

Life-Cycle thinking is the basic idea behind the product-based indicators. Life-Cycle thinking is one of five key principles in the European Union's Integrated Product Policy (IPP) (European Commission, 2003) and is also supported by the United Nations Environmental Programme (UNEP, 2004). In Life-Cycle thinking the cradle-to-grave approach for a product is adopted to reduce its cumulative environmental impacts (European Commission, 2003). The most developed tool for Life-Cycle thinking is Life Cycle Assessment (LCA), which is a method of evaluating a product's resource use and environmental impact throughout its life-cycle. LCA has been used for environmental assessment of milk (Thomassen et al., 2007; Weidema et al., 2007; Thomassen & de Boer, 2005; Cederberg & Mattsson, 2000; Haas et al., 2000), pork (Weidema et al., 2007; Basset-Mens et al., 2006; Dalgaard & Halberg, in prep.;

Cederberg & Flysjö, 2004; Eriksson et al., 2005; Paper 3), beef (Ogino et al., 2007; Weidema et al., 2007), grains (Weidema et al., 1996) and other agricultural/horticultural products (Halberg et al., 2006).

In LCA all relevant emissions and resources used through the life cycle of a product are aggregated and expressed per unit of the considered product. Commonly applied environmental impact categories within LCA of food products are global warming, eutrophication, acidification, photochemical smog and land use. For each of the environmental impact categories the emitted substances throughout the product chain that contribute to the environmental impact category are quantified. For example, when performing an LCA of a livestock product, it is revealed that nitrous oxide is emitted from the soybean production (Paper 3) and methane from the slurry handling (Paper 1). These emissions are standardized and expressed in CO_2-equivalents, thus taking into account that nitrous oxide is a much stronger greenhouse gas than methane and CO_2. Following the same procedure the emissions contributing to the other impact categories are standardized for each of the environmental impact categories. The LCA methodology is standardized according to the ISO-standards (14044), and will be explained in more detail in chapter 2.

Also 'Food miles' and 'Carbon footprint' are product-based environmental indicators that build on life-cycle thinking, and have been used for assessing the environmental impact of food production (Smith et al., 2005; Wiedmann & Minx, 2007). However, these are exclusively focused on global warming and some of them only include fossil CO_2 emissions and not even the two important greenhouse gases methane and nitrous oxide. Therefore, they are considered inadequate for the purpose of this study. However, their use within environmental assessment will be discussed in chapter 4.

1.5 Research questions and outline of the thesis

The overall aim of this thesis is to improve the understanding of the environmental impact of Danish pork production in a global context and to suggest improvements at the most important environmental hotspots. More specifically, the aim is to answer the following questions:

1. Which parts of the product chain of Danish pork are the most polluting?
2. Which substances of those emitted from the product chain of Danish pork are the most polluting?
3. What is the potential of improving the environmental profile of pork by the addition of the feed digestibility improvement factor xylanase to the pig feed or by improved slurry handling?
4. What is the need for methodological improvements within LCA of livestock products?

The outline of the thesis is the following: In chapter 2, I will identify the environmental hotspots of Danish pork, and this is primarily based on results from papers 1, 2 and 3. In chapter 3 options for improvement in the product chain of pork are analysed on basis of the results obtained in papers 4 and 5. In chapter 4, I will discuss the most important methodological aspects regarding LCA of agricultural products, based on my experiences obtained during the Ph.D. period. Then, in chapter 5, I will discuss to what extent the environmental regulation of the Danish pig sector results in environmental improvements at the global and local scale. In chapter 6 the conclusion is presented.

2 Environmental hotspots in the product chain of Danish pork

In this chapter the environmental hotspots in the product chain of pork will be identified by using the product-oriented environmental assessment tool LCA. In this context 'environmental hotspots' refers to the part of the product chain that impacts the environment the most. The presented results are primarily from Paper 3, and before the presentations of the data sources used and the identified environmental hotspots, a short introduction to the LCA methodology and consequential LCA modelling will be given.

2.1 A short introduction to LCA

LCA is chosen as the tool to assess the environmental of pork production and to identify the environmental hotspots in the product chain. The LCA methodology has its own ISO Standard (14044), and has been used for assessing the environmental impact of industrial products for decades (Thrane & Schmidt, 2005). In accordance with the ISO standard, an LCA consists of four interrelated phases, as presented in the subsequent.

The first phase is 'Goal and scope definition', in which the goal of the study and the 'functional unit' are defined. The functional unit is the type and amount of product assessed, and could, for example, be 'one kg pork chops from supermarket'.

The second phase is the 'Life cycle inventory' which involves the compilation and quantification of inputs and outputs in all the involved processes. Outputs include both material outputs (e.g., one kg barley) and emissions (e.g., nitrate leached). In this phase it should also be decided how to handle processes producing more than one product. For instance: Shall the greenhouse gas emissions from the soybean cultivation be ascribed to the soybean meal or to the soy oil? Or should the system be expanded, to avoid this allocation between the two products?

The third phase is the 'Life cycle impact assessment', which is carried out on the basis of the life cycle inventory data. First the emissions in the life cycle inventory data are classified, which means they are assigned to categories according to their impact. For example, methane is a greenhouse gas and is hence assigned to the impact category 'Global warming'. If a substance contributes to more than one impact category, it is assigned to all of them. Classification is followed by characterisation. Every substance is assigned a potential impact in the impact category under study. The potential impact of a substance is given relative to a dominant factor in the category , e.g. for the global warming potential this is typically 1 kg of CO_2 emissions. These relative impacts (the characterisation factors of a substance) are then multiplied with the amount of each emission and the resulting impact values are summed for the respective impact category (Anonymous, 2007c).

The fourth phase is 'Interpretation', where the data from the second and third phases are analysed and conclusions and recommendations are drawn. For example, the environmental hotspots can be revealed in this phase.

For further description on the LCA methodology, see e.g., ISO standard (14044), Thrane & Schmidt (2005), Frederiksen (2004) or Anonymous (2007c).

Two essentially different approaches can be used in LCA modelling: the consequential approach and the traditionally (attributional) approach. As explained in Paper 2, most of the existing LCAs are based on the attributional approach, but the tendency is that the consequential approach is used in an increasing number of studies. In a newer study by Williams et al. (2006) the two approaches are mixed in LCAs of food products. In the present study the consequential approach is used, and will shortly be explained in the following.

The consequential approach has two important main characteristics. The first characteristic is that it seeks to model the technology (or process) actually affected by a change in demand (the marginal technology). This is in contrast to the attributional approach where average (not marginal) technologies are used. An example with electricity is given in Paper 2: In consequential modelling the type of electricity affected by an increased demand is used (coal or gas-based), whereas in the attributional approach the electricity consumption is often modelled as an average of all electricity sources within the region. In consequential modelling the technology (or process) affected is identified by continuously asking: 'what is affected by a change in demand?'. For example: What is affected by a change in demand of Danish pork?

The second characteristic is that co-product allocation is 'systematically' avoided through system expansion (Paper 2). These characteristics are different from attributional LCA, where average technologies (not marginal) are used, and where co-product allocation is often handled by mass or value allocation (Weidema, 2003). An example of the avoidance of allocation is the way the co-production of soybean meal and soy oil is handled in Paper 2. The system expansion, in this case, implies that the inputs and outputs are entirely ascribed to the soybean meal, and the product system is subsequently expanded to include the avoided production of palm oil.

For further details on consequential (and attributional) modelling, see Paper 2, Weidema (2003) and Schmidt (2007). Consequential modelling of agricultural products, its strengths and weaknesses will be discussed in chapter 4. In the following sections the LCA of Danish pork will be presented. The description is divided into four sections in accordance with the four phases of LCA as described earlier. Parts of the description are extracts from Paper 3. See Paper 3 if more detailed explanations are needed.

2.2 Goal and scope definition

The goal was to identify the processes in the product chain of pork with the largest environmental impacts, the so-called environmental hotspots, and thereby the goal was also to answer research questions 1 and 2 in section 1.5. The functional unit was 'one kg Danish pork (carcass weight) delivered to Harwich Harbour in Great Britain'. The one kg of pork must be considered as 'average' pork, with no distinction between the different types of pork (e.g., chop, bacon, tenderloin). There were two reasons for choosing Great Britain as the target destination: i) The transport included both lorry and ship and the study could thereby clarify whether these transport modes were environmental hotspots; ii) Great Britain was the second largest importer of Danish pork, only exceeded by Germany to which the pork only was transported by lorry (Danish Meat Association, 2007). The environmental impact categories considered were global warming, eutrophication, acidification and photochemical smog.

2.3 Life cycle inventory

The life cycle inventory for Danish pork was for the most part established using data from papers 1 and 2. Pig farm and feed grain data were from the National Agricultural Model established in Paper 1 and data on soybean meal were from Paper 2.

The framework for life cycle assessment of Danish pork is presented in Figure 4. Only the most important flows of the product chain of pork are shown. For a more detailed figure and explanations see Paper 3. The most important inputs to the fattening farm were fertiliser (artificial N, P and K), feed, energy (oil for heating and electricity) and weaners. The cash crops sold from the farm (e.g., bread wheat, rape seed and sugar beets) substituted similar products on the market, resulting in an 'avoided production' of agricultural production. Because pig meat was sold from the pig farm, there were co-products from the farm, and the saved emissions due to the avoided production were deducted from the main product (the pork). There were several other avoided products in the product chain of pork that not are shown in figure 4. E.g., the use of soybean meal (an ingredient in the feed) implied co-production of soybean oil, which substituted palm oil (Paper 2), and thus the production of soybean meal resulted in an avoided production of palm oil. The fattening pigs were slaughtered at the slaughterhouse, carved up and accordingly transported by lorry and ship to Harwich Harbour in Great Britain.

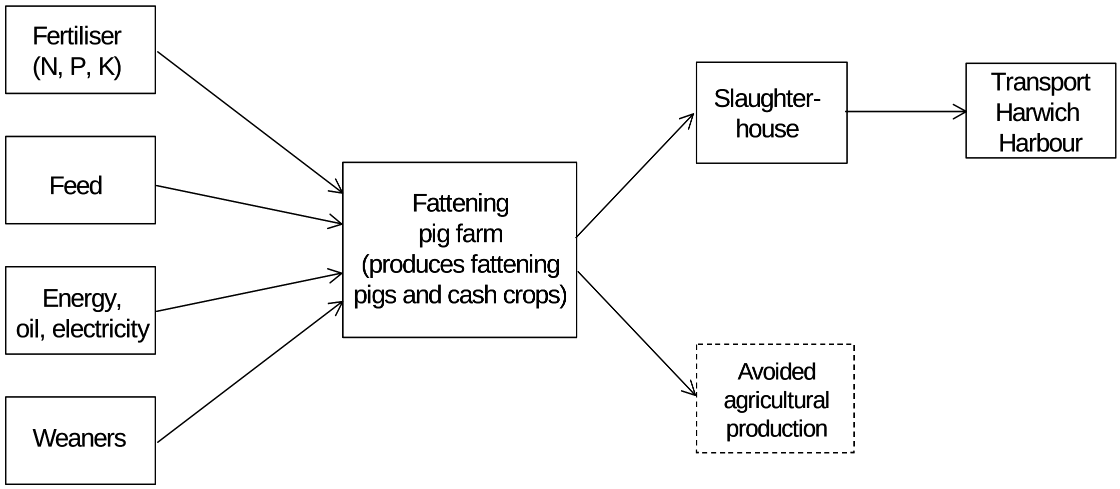

Figure 4. Framework for LCA of pork.

The establishment of the LCA of Danish pork required data on the material flows (e.g., amount of feed, pigs, electricity) and calculation of the emissions from the different parts of the product chain. In the following the pig farm data and the feed data are described. For details on the remaining data (e.g., transport, slaughterhouse) see Paper 3. Finally, the principles used for modelling the emissions from the feed production and pig production are described. For a more detailed description, see papers 1, 2 and 3.

Pig farm data

The pig farm was an environmental hotspot in the product chain of pork (Paper 3), and high quality pig farm data were obviously crucial for the robustness of the LCA results. The LCI of the pig farm used for the environmental assessment of pork in this thesis was from the National Agricultural Model presented in Paper 1. The model was established in order to get data on resource use, production and environmentally important emissions for a set of representative farm types. The 31 modelled farm types were based on farm accountancy data which were representative for the Danish agricultural sector in 1999. The National Agricultural Model contained 31 farm types (6 pig farm types, 8 dairy farm types, 1 sugar beet farm type, etc.), which combined represented the Danish agricultural sector. Identification of the pig farm type that responded the most to a change in demand for Danish pig was achieved using the econometric sector model ESMERALDA (Jensen et al., 2001), as explained in Paper 3.

The pig farm used for the LCA of pork produced 1402 fattening pigs per year. The weaners arrived at the farm at 30 kg and were taken to the slaughterhouse when they obtained a weight of 100 kg. The farm also sold rape seed, bread wheat, sugar beets, straw, peas and grass seeds and had a farmed area of 71 hectares, of which 68% was devoted to grain production. The self-sufficiency in feed (measured in N) was 41%, meaning that 59% of the feed consumed by the fattening pigs was purchased. The feed produced on the farm was exclusively grain (barley and wheat).

An important strength of deriving data from the National Agricultural Model was that the farm types were representative, partly because of the use of the representative data set of farm accounts and partly because of an adjustment to national level statistics. The farm types were based on realistic and documented levels of resource use per unit agricultural product and the emissions, therefore, reflected average production levels and efficiency within different farm types. The farm types were all consistent in terms of crop-livestock interactions, and together they formed the National Agricultural Model that documented the total resource use and emissions of the Danish agricultural sector in 1999, including the exchange of slurry and straw between farm types (Paper 1). There can be large differences in the environmental impact per kg pork produced at different farms as shown by Hvid et al. (2005) and therefore it was preferable to use representative farm data instead of case farm data. For further details on the establishment of the National Agricultural Model, see Paper 1.

Feed data

When more pigs are produced, more pig feed will be required on the world market. However, only the most competitive pig feed ingredients on the world market will be affected. The most competitive protein meal on the world market is soybean meal as argued in Paper 2, and the most competitive energy source is grain (Paper 3). Hence, when using the consequential modelling approach it was sufficient to have LCA-data on soybean meal and feed grain. For further explanations, see papers 2 and 3.

LCA-data on soybean meal were from Paper 2. Soybeans contain protein (approx. 35%) which is used for livestock feed after crushing and extraction of the oil. The oil constitutes approx. 18% of the soybeans and is primarily used for consumption. Consequential modelling is applied for the LCA of soybean meal and because the soybean meal has the co-product soybean oil, the avoided production of the most competitive oil (palm oil) is included in the calculations. The soybeans from Paper 2 were cultivated and processed in Argentina, transported by lorry to Rosario Harbour in Argentina, and finally shipped to Rotterdam Harbour in the Netherlands. For further details, see Paper 2. In the LCA of Danish pork (Paper 3) the soybean meal transport by lorry from Rotterdam Harbour to Denmark was added.

LCA-data on feed grain were from the LCAfood-database (www.LCAfood.dk), and were based on the same farm account data (Paper 1) as the pig farm LCI described in the previous section. The feed grain was a mix of spring barley (25%), winter barley (25%) and wheat (50%) produced in Denmark. Schmidt (2007) argues that grain produced in Canada is the most competitive on the market, and thereby the marginal grain. But in this study I preferred to use high quality data from the wrong country (Denmark) instead of low quality data from the right country (Canada).

Quantification of emissions from agricultural production

The same principles of quantifying the emissions from the agricultural production (pigs, soybean meal and feed grain) were used in papers 1, 2 and 3. In all three papers the nutrient balance approach (Kristensen et al. 2005a: Halberg et al., 1995) was applied, following the framework presented in figure 5. The nutrient balance approach can be applied to agricultural systems at various levels: animal level, herd level, field level and farm level. Figure 1 is an example of how the nutrient balance approach had been applied to the pig farm type, which was established in Paper 1 and used for environmental assessment of pork in Paper 3. The methodology used for modelling the inputs (e.g., feed, fertiliser, diesel) to the farm and outputs from the farm (e.g., pigs, rape seed) is described in Paper 1. The upper part of figure 1 shows the framework for modelling the N emissions from the pig farm. Firstly N inputs were calculated, based on knowledge of N content in the feed (Møller et al., 2003) and amount of artificial fertiliser-N used. Secondly, N in the outputs was calculated based on knowledge of N content in the cash crops sold from the farm (Møller et al., 2003) and N content in the pigs sold from the farm (Poulsen et al., 2001). The N surplus was then calculated by subtracting the N output from the N input. The N surplus was the N imported to the farm but not incorporated in any of the products. The N surplus will be lost to the environment in different forms as shown on the right in figure 5, and the procedure used for dividing the N surplus into the different pathways of loss is described in the following. Ammonia is volatilized into the air, primarily from the slurry, but also from artificial fertiliser and growing crops. The amount of ammonia emitted from the pig farm was calculated based on the amount of slurry and artificial fertiliser, crop types and ammonia emission factors. Ammonia contributes to both eutrophication and acidification potential, and also impacts the biodiversity. Some of the N surplus is lost by denitrification, which includes all the microbiological processes that are converting N in the soil and slurry to N_2 or nitrous oxide (N_2O) (Vinther & Hansen, 2004). Also other N substances can be formed during denitrification, but N_2 and N_2O are the most important. N_2 is harmless to the environment, and the N_2 concentration in the atmosphere is 75.5 w-% (Helt & Rancke-Madsen, 1991). The N lost by denitrification was calculated according to Vinther & Hansen (2004), as described in Paper 1. Part of the N surplus is incorporated in the soil or the N is released from the soil, depending on the way the soil is cultivated. The nitrate leaching was then assumed to be equal to the N surplus minus ammonia losses, denitrification and N change in soil N status. Nitrate contributes to the eutrophication potential.

Nitrous oxide emissions were calculated according to the IPCC methodology (IPCC, 2000), and therefore required data on amounts of N in slurry, fertiliser applied to the soil, ammonia losses, crop residues and nitrate leached. All these flows were quantified in order to calculate the N surplus, and were therefore available. Using these N flows also ensured consistency between the estimated N losses (ammonia, denitrification, net change in N in soil, nitrate leaching shown in figure 5) and the estimated nitrous oxide emissions. This procedure was used in both Paper 1 and Paper 2. Methane emission from slurry handling and storage was calculated according to IPCC (2000), as explained in Paper 1.

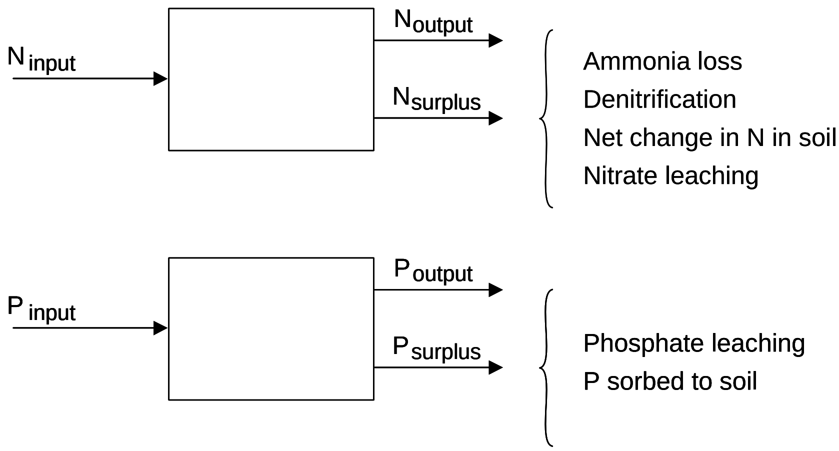

Figure 5. Nutrient balance approach used in papers 1, 2 and 3 for calculation of N and P emissions. Input - Output = Surplus.

Emissions of phosphate to the aquatic environment were also calculated by use of the nutrient balance approach, as shown in the lower part of figure 5. However, the fate of P surplus in the environment is more simple compared to N. P surplus can be lost as phosphate to the aquatic environment or sorbed to the soil. The proportion of P surplus leached as phosphate depends highly on the soil characteristics, climate, topography of the field, etc. According to Poulsen & Rubæk (2005) 1000 t P was leached in Denmark in year 2000, and combining this value with the total P surplus (34,000 t P) from all the farm types in the National Agricultural Model (Paper 1), it was assessed that 2.9% of the P-surplus was leached as phosphate. Thus it was assumed that the P surplus was directly proportional to phosphate leaching. The same assumptions were used for the establishment of the life cycle inventory for soybean cultivation in Paper 2 (table 2). Some of the assumptions presented above might be controversial, and will therefore be discussed in chapter 4.

An important strength of the nutrient balance approach is that all N and P is accounted for, hence securing consistency between the different types of N and P losses. All N and P entering the farm (or animal, or field) will leave as products or emissions. A weakness is the level of uncertainty for nitrate leaching , because nitrate is calculated as N-surplus minus ammonia, denitrification and net change in soil N status (figure 5), and the uncertainties will therefore be summed in the estimate on nitrate leaching. However, the total amount of nitrate leached from The National Agricultural Model (Paper 1) was in good agreement with results on nitrate leaching obtained in the 'Evaluation of the Action Plan for the Aquatic Environment II' (Paper 1). This emphasizes the robustness of the methodology used for calculating nitrate leaching in this study.

2.4 Life cycle impact assessment

Several methods are available for Life Cycle Impact Assessment (LCIA). The EDIP97 (Wenzel et al. 1997, updated version 2.3) was used in the LCA of soybean meal (Paper 2) and in the LCA of Danish pork (Paper 3), while Ecoindicator 95 was used in Paper 4.

The Intergovernmental Panel on Climate Change (IPCC, 2001, Chapter 6) has launched new, but lower, characterization factors for nitrous oxides (296 g CO_2/g) and methane (23 g CO_2/g). A sensitivity analysis was performed and it was found that the substitution of the EDIP97 characterization factors with IPCC (2001) characterization factors reduced the global warming potential by 6.1% for pork and 3.7% for soybean meal. The changes were small and although some of the environmental hotspots in the product chain of pork became somewhat 'hotter', this was deemed unlikely to change the conclusions of this thesis, thus the EDIP97 was used for the LCIA.

2.5 Interpretation

The characterized results per kg Danish pork delivered to Harwich Harbour were 3.77 kg CO_2 eq. global warming potential, 319 g NO_3 eq. eutrophication potential, 59 g SO_2 eq. acidification potential, and 1.27 g ethene eq. photochemical smog potential, as presented in Paper 3 (table 3). These results were comparable to LCA results of pork produced in Sweden (Cederberg & Flysjö, 2004) and France (Basset-Mens & van der Werf, 2005). For further details on the comparisons, see Paper 3 (table 5). In a new study (Dalgaard & Halberg, in prep.) on LCA of Danish pork produced in 2005, the characterized results were lower (3.6 kg CO_2 global warming potential, 232 g NO_3 eq. eutrophication potential and 45 g SO_2 eq. acidification potential) than the results in this study. There are two important reasons for this discrepancy. Firstly the feed efficiency for both fattening pigs and weaners has improved from 1999 to 2005 (Sloyan et al., 2006), hence less pig feed is used per kg pork produced. Secondly, the new characterization factors on global warming from IPCC (2001) and a new IPCC methodology for estimating the greenhouse gas emissions from the agricultural sector (Eggleston et al. 2006) were used in the study of Dalgaard & Halberg (in prep.).

According to Williams et al. (2006) the environmental impact per kg pig was much higher than the results of our study and the studies from Sweden and France (Cederberg & Flysjö, 2004; Basset-Mens & van der Werf, 2005). It would appear that the main difference between our results and the results of Williams et al. (2006) relates to the method for calculating nitrous oxides. In addition, much more ammonia was emitted per kg pig and presumably the nitrate leaching from the soybean used in the study of Williams et al. (2006) was higher compared to the soybean meal which we have used (Paper 2).

The contribution from the different stages of the pork's product chain to the respective environmental impact categories is presented in figure 6.

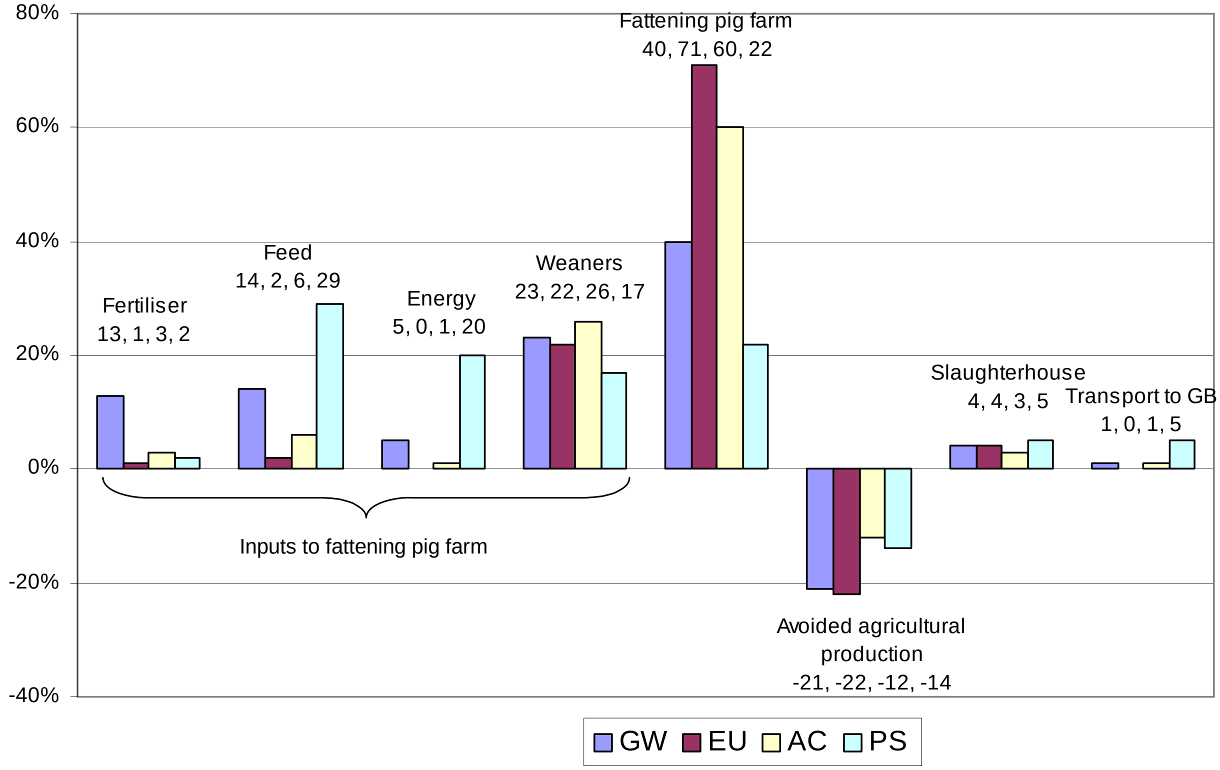

Figure 6. Contribution to global warming potential (GW), eutrophication potential (EU), acidification potential (AC) and photochemical smog potential (PS) from the eight stages of pork's product chain. The y-axis shows the percentages of emissions arising from the different stages of the product chain, and the contribution in percentages (from each of the life cycle stages to the respective environmental impact categories) is shown below the name.

The eight stages in the product chain presented in figure 6 correspond to the eight boxes shown in figure 4. 'Fertiliser', 'feed', 'energy' and weaners are solely the farm inputs purchased to the fattening pig farm. Similar farm inputs were used at the weaner farm, but the environmental impacts related to the farm inputs used in the weaner production were included in 'weaners'. 'Fertiliser' includes artificial fertiliser (N, P and K). 'Energy' includes electricity, and processing and distribution of oil used for heating and diesel for traction. The emissions from 'fattening pig farm' include all emissions from the housing and the 71 hectares of agricultural land, although part of the agricultural land was cultivated with cash crops. The crops sold from the 'fattening pig farm' substituted other products on the market, which resulted in 'saved' emissions. These 'saved' emissions are represented by the negative values called 'avoided agricultural production'. The y-axis shows the percentages of emissions that arise from the different stages of the product chain, and the contribution in percentages (from each of the life cycle stages to the respective environmental impact categories) is shown below the name. For example, 13% of the greenhouse gases emitted (measured in CO_2-eq.) from the product chain of pork came from the production and distribution of artificial fertiliser. If the other positive contributions (feed (14%), energy (5%), weaners (23%), fattening pig farm (40%), slaughterhouse (4%) and transport to Great Britain (1%)) are added they sum up to 100%. 21% of the greenhouse gas emissions were counterbalanced by ‘avoided agricultural production’.

The fattening pig farm and the weaners were the two most important environmental hotspots for both global warming, eutrophication and acidification, whereas the feed imported to the fattening pig farm, the energy use and the fattening pig farm itself were the most important environmental hotspots seen in relation to photochemical smog. In the following the contribution from each of the stages in the product chain of pork will be described in more detail.

Global warming potential

As described in Paper 3, the main contributors to the global warming potential were the fattening pig farm (40%) and the weaner farm (23%). Of the greenhouse gases emitted on the fattening pig farm 72% was nitrous oxide (figure 7), whereas fossil CO_2 from the use of agricultural machinery only contributed 11%, and methane from slurry handling contributed 17%. The different sources of nitrous oxide emissions at the fattening pig farm are presented in figure 8, and it shows that 43% of the nitrous oxide came from denitrification of nitrate leached from the field, and 44% originated from denitrification of N fertiliser (artificial fertiliser and slurry) applied to the fields.

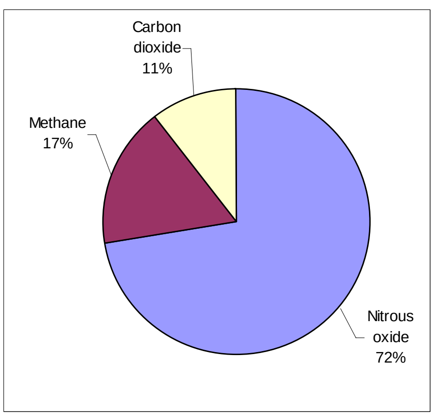

Figure 7. Types of greenhouse gases emitted from the fattening pig farm. Unit: CO_2-eq

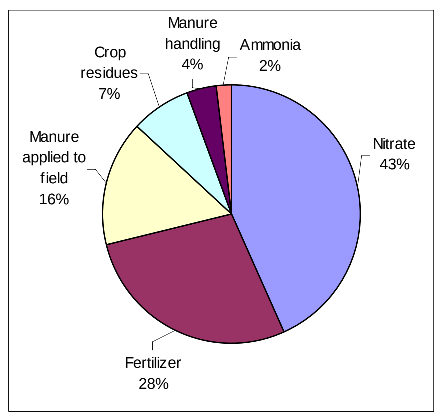

Figure 8. Sources of nitrous oxide emissions on the fattening pig farm.

Figure 6 shows that production and distribution of fertiliser imported to the fattening pig farm contributes 13% of the greenhouse gases emitted. Of this, 94% is related to the production of artificial fertiliser-N and only 6% to the production of P and K (not shown), because these fertiliser types were used in smaller amounts and less energy was used for their manufacturing. Of the emitted greenhouse gases, 14% was related to the production of feeds imported to the fattening pig farm, whereof 79% could be ascribed to production and distribution of soybean meal. So besides the fattening farm and weaner production, production and distribution of soybean meal and artificial fertiliser-N can also be considered as being environmental hotspots. The contribution from ‘slaughterhouse’ (which includes transport from farm to slaughterhouse) was only 4%, while the transport from the slaughterhouse to Harwich Harbour only contributed 1%. However, much of the exported Danish pork is transported far longer distances than to Great Britain, and to explore the impact of the transport distance on the global warming potential per kg pork, two additional scenarios were established. One where the pork was transported to Munich in the south of Germany by lorry (distance: 1075 km) and one where it was transported to Tokyo Harbour in Japan by ship (distance: 21,153 km). The transport to Munich and Tokyo increased the global warming potential by 3% and 5%, respectively. This shows that the contribution from transport is limited, although the pork is transported long distance. ‘Food miles’ is a measure of the distance food travels from the farm to the consumer (Smith et al., 2005) as explained in chapter 1, and the low contribution from the transport of pork in this study highlights that ‘Food miles’ is a concept that cannot stand alone as an indicator of environmental impact from food production. CO_2 emitted from the transport of the pork is not an environmental hotspot, and even if the food miles of pork were dramatically reduced, it would hardly have any effect on the emissions of greenhouse gases. This will be discussed in more detail in chapter 4.

Considering the product chain of pork, the substances with the highest contribution to global warming potential were nitrous oxide (53%), CO_2 (37%) and methane (10%). If the lower characterization factors for nitrous oxide and methane (IPCC, 2001) (see section 2.4) are applied, the contribution will only be slightly lower for nitrous oxide (51%) and slightly higher for CO_2 (39%). Consequently, there is no reason to believe that the environmental hotspots regarding global warming potential would change if the new characterization factors from IPCC (2001) were used.

Eutrophication potential

According to Paper 3, the contribution to eutrophication was 71% from the fattening pig farm and 22% from the ‘weaners’. Out of this, 69%, 28% and 3% came from nitrate, ammonia and phosphate, respectively. Nitrate and phosphate were leached from the fields. Ammonia was primarily emitted from the animal house, during storage in slurry tanks and under and after application of the slurry to fields. Considering the whole product chain of pork, the two largest contributors were nitrate (63%) and ammonia (30%). The contribution from phosphate was only 3%, and this might lead to the conclusion that phosphate leaching from pig production is not an issue. However, this conclusion is at variance with the ‘Action Plan for the Aquatic Environment III’ (Anonymous, 2004) in which reduction of phosphate leaching from the agricultural sector is given high priority. So although phosphate had not been identified as an important contributor to eutrophication potential in the present LCA study, it cannot be neglected. This will be discussed further in the chapter 4.

Acidification potential

The fattening pig farm and the production of weaners contributed 60% and 26%, respectively, of the emitted acidifying substances. Ammonia from the farms amounted to 83% of the acidifying substances emitted from the product chain of Danish pork. The ammonia emitted from the fattening pig farm came from slurry in the animal house (38%), storage of slurry in slurry tanks (12%), application of slurry (20%) and N fertiliser (11%) to the fields and from the crops (19%). ‘Feed’ accounted for 6% of the acidifying substances emitted, and fertiliser and ‘energy’ purchased by the fattening pig farm accounted for 3% and 1%, respectively. The contributions from ‘slaughterhouse’ and ‘transport to UK’ were 3% and 1%, respectively, primarily related to energy use (Paper 3). Ammonia contributed 84% of the emissions form the product chain and was thereby the largest contributor.

Photochemical smog potential

Substances contributing to photochemical smog primarily came from refining and combustion of fossil fuel. ‘Feed’ was the largest contributor (29%), with soybean meal being more important than feed grains. ‘Fattening pig farm’ contributed 22% of the photochemical smog potential, and out of this 79% could be ascribed to ‘Non-Methane Volatile Organic Compounds’ (NMVOC) deriving from the fossil fuel. Methane emitted from the slurry also contributed to photochemical smog potential. Considering the entire product chain of pork, the major contributor was NMVOCs, which contributed 86%.

3 Options for improvements in the product chain of pork

As described in the previous chapter the N compounds (nitrate, ammonia and nitrous oxide) are important pollutants, and the emissions of these pollutants must be reduced if the goal is to improve the environmental profile of pork. There are several ways of reducing these emissions (Paper 3, Grant & Waagepetersen, 2003; Oenema et al., 2006; Taminga 2003). For example the addition of digestibility-improving enzymes to the feed can improve the absorption of e.g., energy and protein, hence decreasing the amount of feed required for producing an equal amount of meat, as explained in Paper 4. Another option is to change the slurry handling by for example anaerobic digestion or separation of the slurry, as shown in Paper 5.

In the following two sections I will analyse how and to what extent the environmental profile of pork can be improved by addition of digestibility-improving enzymes to the feed and by using slurry technologies. Obviously, several other options for improving the environmental profile of pork exist, but these were outside the scope of the present study.

3.1 Improving the digestibility of the feed

Feed and slurry production are important environmental hotspots in the product chain of pork, as stated in the previous chapter. The feed digestibility is a key parameter for achieving reductions in both the feed consumption and slurry excretion per pig produced. The feed digestibility, in terms of kg feed consumed per kg pig produced, was on average 2.67 for Danish fatteners in 2005, and is thus 3% better compared to 2002 and the second best within Europe, only exceeded by the Netherlands (Sloyan et al., 2006). Means of improving the feed digestibility are, for example, more accurate feeding or the addition of enzymes to the feed. In Denmark the enzymes xylanase and phytase are often applied to pig feed.

Xylanase improves the digestibility of nutrients (proteins) and energy (Tybirk, 2005b; Moehn et al., 2007), whereas phytase improves the digestibility of P (Tybirk, 2005a). In chapter 2 it was concluded that the N compounds nitrate, ammonia and nitrous oxides played major roles. But to what extent can xylanase contribute to improvement of pork's environmental profile?

In Paper 4 an environmental assessment of xylanase was conducted, using the consequential LCA approach. Xylanase is widely used in the pig sector and has penetrated about 30% of the feed market in Europe (Paper 4), and Tybirk (2005b) estimated that fattening pigs can be fed 3% less feed and still produce the same amount of meat if xylanase is added to the feed. In chapter 2 it was stated that the feed production was an environmental hotspot and therefore reduction of feed consumption per pig will obviously be environmentally beneficial.

In Paper 4 it was assumed that addition of xylanase to the feed reduced the pigs' feed demand by 2.5% without reducing the amount of meat produced. Furthermore it was assumed that the feed with xylanase had a lower soybean meal and fat content, but a higher content of barley. These assumptions were based on modelling with software used in practical animal feed optimisation procedures. Further details regarding methodology and data sources are explained in Paper 4.

Less protein consumed per unit pig produced resulted in less excretion of N in slurry, and thus less ammonia, nitrous oxide and nitrate was emitted from the pig housing, the slurry storage and from the fields. However, the lower N content in the slurry resulted in a smaller replacement of artificial fertiliser, which again resulted in more use, transport and production of artificial fertiliser and hence an increased environmental impact. The manufacturing and transport of the enzyme xylanase also had an effect on the environment (Nielsen et al., 2007). Nevertheless, the environmental advantages of the use of xylanase (reductions in feed consumption and slurry production) were, in general, larger than the disadvantages (increased production and transport of xylanase and artificial fertiliser). So, according to Paper 4, it can be concluded that in environmental terms the negative aspects of xylanase did not cancel out the positive aspects.

But to what extent can the environmental profile of pork be improved? If results from Paper 3 are combined with the results from Paper 4, a rough estimate can be given. In Paper 4 the functional unit was not one kg pig meat as in Paper 3. The scope in Paper 4 was to provide an assessment of the changes in environmental impacts when switching from producing one tonne of feed without xylanase to a nutritionally equivalent (but reduced quantity) of feed. Therefore, only emissions that were affected by the change in feed were quantified and included in the calculations. For example, figure 3 (in Paper 4) showed that 77 kg CO_2-eq. (78-0.6) could be saved if one tonne feed without xylanase was substituted with feed with xylanase. This corresponds approximately to 0.185 kg CO_2-eqs. per kg meat. According to Paper 3, the global warming potential was 3.5 kg CO_2-eq. per kg pork (carcass weight from farm gate (table 5 in Paper 3)). Consequently, combining the results from papers 3 and 4, it appeared that the global warming potential per kg pork could be reduced by approximately 5%, with an addition of xylanase to the pig feed. This reduction in global warming due to addition of xylanase was to a large extent driven by the reduced use of soybean meal (reduced emissions of nitrous oxide from soybean fields). However, the reduction in eutrophication potential induced by application of xylanase is less than 1% and therefore very limited.

It should be noted that EcoIndicator 95 (version 2.03) was used in the assessment of the digestibility-improvement factor xylanase in Paper 4. The characterization factors for the two important greenhouse gases (nitrous oxide and methane) are lower in EcoIndicator 95 compared to EDIP97. Nitrous oxide is 270 (EDIP97) versus 320 g CO_2/g (EcoIndicator), while methane is 11 versus 25 g CO_2/g. In order to secure consistent comparisons, the results form Paper 4 were characterized by using EDIP 97, before they were combined with the results form Paper 3.

In a recently published study by Moehn et al. (2007) a tendency for increased methane production from xylanase-supplemented diets was observed. Methane is a greenhouse gas and therefore some of the reduction in greenhouse gas emissions due to the use of xylanase (Paper 4) might be partly counterbalanced.

Another digestibility-improving enzyme is phytase. Phytase application to the pig feed improves the P digestibility (Tybirk, 2005a; Moehn et al., 2007), and therefore less mineral P can be added to the feed and less P will be excreted in the slurry. A study by Nielsen & Wenzel (2006), who used the consequential LCA approach, showed that if mineral feed was applied to the feed, the global warming, acidification and nutrient enrichment potentials would be respectively 17, 110 and 700 times higher compared to a situation where phytase was applied to the feed . The production of mineral P feed required considerably more energy than phytase did and this was the reason for the lower global warming potential when phytase was used. But according to Paper 3 the contribution to global warming potential from the production of mineral P for feed is only 0.44% and is therefore not even presented in Paper 3. So if the global warming potential per kg pork is to be reduced, the most obvious place to start is not with the use of phytase. However, the effect of phytase on reduced phosphate leaching cannot be ignored.

These two examples with xylanase and phytase show how the addition of enzymes to the feed can improve the digestibility of the feed and how this to a larger or smaller extent can effect the environment. Improved digestibility can also be obtained without the use of enzymes, and just with improved farm management. Another way of improving the environmental profile of pork is to optimize the handling of slurry, as it will be explained in the following.

3.2 Improving the slurry handling

Slurry was identified as an environmental hotspot in the product chain of pork (Paper 3), and it contributed to both global warming (because of nitrous oxide), acidification (because of ammonia) and eutrophication (because of nitrate, ammonia and phosphate). Moreover, in Paper 4, it was shown how the environmental impacts of slurry could be reduced if xylanase was applied and thus reducing the N content in the slurry. So slurry is a major player in the pig sector's impact on the environment.

Different types of slurry technologies exist, and they are increasingly used on Danish pig farms (Birkmose, 2007). One of the technologies is slurry separation, and its potential for environmental improvements was analysed in Paper 5. Slurry separation is a technology where the raw slurry is separated in a decanter centrifuge, which separates the solids from the liquid. The solid fraction (from now on designated the 'fibrous fraction') contains straw and fibre from pig dung and pig hairs, while the liquid fraction contains most of the water and pig urine. The liquid fraction has high N and low P contents compared to the fibrous fraction (Paper 5).

In Paper 5 it was concluded that if slurry separation was used on pig farms with a high livestock density and the fibrous fraction was exported from the pig farms, it had the potential of reducing the environmental impact. The amount of slurry transported could be reduced by 37% and the amount of P applied to the fields at the pig farm could be reduced by 82% compared to a situation where no slurry separation was performed. However, these environmental improvements required that the slurry separation plant separated efficiently, and this was not the case at the two private pig farms from where data were collected. The separation efficiencies were low at these farms, resulting in an only 25% reduction in the amount of P applied per hectare at the pig farm.

Slurry separation is a newer technology than anaerobic digestion and obviously there are environmental problems that need to be solved. For instance, 22% of the fibrous fraction produced from slurry separation in Denmark has no purchaser (Birkmose, 2007). Presumably many of the cash crop farms that could import the slurry find it easier and safer to purchase artificial fertiliser. Thus the environmental advantage of cash crop farmers using less artificial P fertiliser is lost. Companies producing artificial fertiliser (e.g., Kemira Grow How, DLG) have tried to convert the fibrous fraction to a product that resembles artificial fertiliser, with the intention of selling this product to farmers who normally purchase artificial fertiliser (Hinge, 2003). However, the cost of the processing is so high that it not is profitable for the companies to process the fibrous fraction (Hinge, 2003). The costs of processing the fibrous fraction could presumably be reduced if more of it was produced, but at the same time problems with disposal of a large fibrous fraction may be a barrier for farmers who contemplate investing in slurry separation plants. This is obviously a dilemma. Another problem regarding the fibrous fraction is that there might be substantial losses of N and carbon when heaped. Petersen & Sørensen (submitted) quantified N and carbon losses from fibrous fraction heaps at the two private farms studied in Paper 5. They found that the losses of ammonium-N, total-N and carbon during storage of the fibrous fraction were in the range 30-90%, 10-55% and 35-70% of the initial amount, respectively. A significant proportion of the N losses was assumed to be ammonia, and part of it might have been nitrous oxide, whereas the C losses were in the form of CO_2 or methane. This means that from the storage of the fibrous fractions at pig farms there is a risk of emission of pollutants that contribute both to global warming, acidification and eutrophication potentials. However, the level of these emissions is not known, and it is therefore difficult to assess how important these emissions are in relation to the other parts in the product chain of pork.

In Paper 5 it was furthermore concluded that if biogas from anaerobically digested slurry was used for heat and power production and thus substituting fossil energy, considerable amounts of greenhouse gases could be saved. It was shown that if all the slurry from a pig was used for energy production (and accordingly as fertiliser) the greenhouse gas emission per kg pork (carcass weight) could be reduced by 16%. The potential for reducing the greenhouse gas emissions was many times higher if the slurry was anaerobically digested compared to a situation where it was separated. So seen in relation to global warming, anaerobic digestion was not only better than slurry separation, but it also offered the opportunity to reduce the global warming potential per pig considerably. On the other hand, anaerobic digestion did not change the nutrient contents (and not the P/N ratio) in the slurry, so it did not have the same potential for reducing the P loads on the pig farm that slurry separation had (Paper 5)

4 Considerations regarding LCA of agricultural products

The purpose of this chapter is to discuss the strengths and weaknesses of LCA. First I will compare LCA to other environmental indicators, and then discuss some aspects on the use of consequential modelling. Finally, the weak points of quantification of pollutants in LCA will be highlighted.

4.1 LCA in comparison with other environmental indicators

A distinction between the area-based and product-based environmental indicators was made in chapter 1, and the product-based environmental indicators 'Food miles' and 'Carbon footprint' were mentioned. 'Food miles' is a term which refers to the distance food travels from the farm to the consumer (Smith et al., 2005) and is used as an environmental indicator for food products. It has gained much attention in the debate, especially in Great Britain (e.g., Smith et al., 2005). 'Food miles' is used not only as an indicator of environmental sustainability, but also of economic and social sustainability (Smith et al., 2005). But the question is to what extent reduction in food miles will increase the environmental sustainability. In this study it was shown that the contributions from transport of pork to the global warming potential were 1%, 3% and 5% if the pork was transported from Horsens Slaughterhouse in Denmark to Harwich (Great Britain), Munich (Germany) and Tokyo (Japan), respectively. A similar conclusion was made for soybean meal in Paper 2. Although the soybean was transported from Argentina to the Netherlands, the contribution to global warming potential was higher from the soybean cultivation (primarily because of nitrous oxide) than from the transport. Moreover, the contributions from lorry (transport of soybeans in Argentina (500 km)) and ship (shipping of soybean meal from Argentina to Rotterdam in the Netherlands (12,082 km)) were very similar, despite the large differences in distances. Thus the contribution from transport to global warming was low compared to the contribution form the pig farms and the fields mainly because nitrous oxide is a very strong greenhouse gas and emitted in considerable amounts. Moreover, ‘Food miles’ can be misleading because the transport mode (ship, aircraft or lorry) is often not included in the calculations. For example, transport by lorry emits considerably more CO_2 than transport by ship (EcoInvent Centre, 2004). So if ‘Food miles’ is to be used as an environmental indicator, it is crucial to divide it into mode of transport, for example, ‘ship food miles’ and ‘lorry food miles’. ‘Food miles’ is an inadequate environmental indicator. If the attention is to reduce the greenhouse gas emissions, focus should not be set on reduction of food miles, but on the real environmental hotspots.

‘Carbon footprint’ is another environmental indicator, which is used in various forms (Wiedmann & Minx, 2007), and it must be used with care if applied to food products. If nitrous oxides and methane are not included in the calculation, the food product’s impact on global warming will be underestimated and comparison of products might give a misleading result.

However, the environmental indicators ‘Food miles’ and ‘Carbon footprint’ have one large advantage: they are much easier to communicate to people who have no knowledge of environmental issues. The term ‘Food miles’ has associations of polluting aircrafts or ships and the term ‘Carbon footprint’ has associations of a footprint that harms the nature and therefore should be minimized. Both terms are easy to visualize for the consumers and have a ‘feel good factor’. In contrast, the term ‘Life Cycle Assessment’ is difficult to visualize and does not necessarily guide the thoughts to environmental issues. Understanding the LCA methodology is not straightforward for people without environmental knowledge, primarily, because LCA operates with several environmental impact categories (e.g, acidification, toxicity, global warming) and units (e.g., SO_2 equivalents, person equivalents) that are more difficult to understand. Moreover, an LCA-based comparison of two products will often conclude that product A in comparison with product B, is better in one impact category, but worse in another. Most people will presumably prefer a clearer judgement, and this can be provided by ‘Food miles’ and ‘Carbon footprint’, because they only operate with global warming.

Nevertheless, an important quality in LCA is that it offers the opportunity of assessing several types of environmental impacts (acidification, global warming etc) for a product. It makes it easier to assess whether mitigation of one type of emission implies an increase in other types of emission. An example could be acidification of slurry which reduces ammonia emission

(Kai et al., 2007). If only one environmental indicator, e.g. ammonia loss per animal, is calculated, the slurry acidification will be defined as ‘beneficial for the environment’. But maybe acidification of slurry would not be beneficial for the environment if the contribution to eutrophication and global warming was also assessed. What is, for example, the environmental impact of producing the acid? Will more nitrate be leached from the fields because the slurry contains more N due to reduction of ammonia losses? Will more greenhouse gases be emitted, because the lime production is increased in order to raise pH in the soil? Agricultural systems are complicated and there is a large danger of pollution swapping if decisions are taken on too narrow a basis. Holistic environmental indicators, as those established by LCA, must be the basis in order to prevent undesired effects.

In Paper 1 (table 6), several area-based environmental indicators (N-surplus per ha, P-surplus per ha, ammonia emissions per ha, etc.) were presented, and it should be emphasized that these environmental indicators are still usable and should not be substituted by LCA-based environmental indicators. For regions with intensive agricultural production or vulnerable natural habitats it is important to know the amounts of pollutants emitted in that specific area, and not very relevant whether the production results in e.g. phosphate leaching in other parts of the world. On the other hand, area-based indicators cannot stand alone if greenhouse gas emissions are to be reduced. Area-based and LCA-based indicators can supplement each other in identifying the environmental hotspots, both in a local and a global context.

An obvious strength of LCA is that the methodology is well developed, it has been used for decades, much software with data bases and life cycle impact assessment methods is available, and the methodology is ISO standardized. However, although LCA is well described, there are still two very distinct ways of modelling an LCA – the attributional and the consequential approach, as described in chapter 2. In this thesis I have applied the consequential approach and in the following section some of the aspects regarding the use of consequential LCA will be discussed.

4.2 Consequential LCA

In chapter 2 (section 2.1) two important main characteristics of the consequential approach were described. The first was how the consequential approach seeks to model the technology (or process) actually affected by a change in demand, and the second was how system expansion is used to avoid allocation. In the following some examples of how these characteristics are applied in this study are presented. Furthermore, the advantages and disadvantages of consequential modelling will be discussed.

Example 1: Incorporation of slurry exchange between farms in LCA

In this example it will be described how the consequential approach has been used to avoid allocation and to include the negative and positive aspects of slurry production. To a large extent the text is similar to a paper (Dalgaard & Halberg, 2007) I presented at the ‘5th International Conference ‘LCA in foods’ in Sweden’.

Slurry affects the environment negatively because it causes emissions of ammonia, nitrous oxide, nitrate and phosphate, both during storage and when the slurry is applied as fertiliser to field-grown crops. On the other hand, slurry might also contribute positively to the environment, if it substitutes artificial fertiliser or is used for energy production and thus substitutes fossil fuel (Paper 5). In an integrated farming system where slurry is recycled to feed crops only, it does not matter whether slurry emissions are allocated to the pigs or the feed crops, since the environmental burden will be allocated to the pigs in any case. But when slurry is used in cash crop production, whether on the pig farm itself or after export to another farm, then the question of allocation of emissions from handling slurry arises. In order to facilitate comparisons of LCAs on agricultural products it is important to have clear and transparent methods. Using the consequential modelling principles a framework for handling of slurry in LCA is presented.

The objective is to establish a framework for handling livestock slurry in LCA, and thereby give answers to the following question: How to account for emissions from slurry in an LCA of livestock products? Shall the environmental impact from slurry be ascribed to the pig or the cash crops to which the slurry is applied? Following the consequential methodology the starting point is ‘What is affected by a change in demand? In this context it can be stated that an increased demand for pork results in an increased production of slurry. The emissions from this extra slurry are logically an environmental burden on the pork regardless of whether it is used on the farm itself or exported. Consequently, all the extra emissions arising from using the pig slurry in cash crop production should “burden” the environmental profile of the livestock products. On the other hand, this environmental cost should be deducted any saved emissions arising in the cash crop production from replaced fertiliser. Thus, the principles of using systems expansion for handling co-products in LCA are followed (Weidema, 2003). Consequential LCA modelling was therefore performed, which included the slurry related emissions on the cash crop farm and the avoided production of artificial fertiliser. Calculation of the emissions from pig housing, storage and field was based on Paper 1. The amount of avoided artificial fertiliser is based on data from the Danish Environmental regulation. The Danish regulation stipulated that for each 100 kg of N applied in pig slurry to a crop the fertiliser should be reduced by 60 kg N compared to the public norm for the particular crop on the particular soil type.

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

The inventory and characterized results per kg slurry-N exported from a pig farm to a cash crop farm are presented in table 1. Each kg slurry-N exported from the farm results in an avoided production of 600 g artificial fertiliser-N, and extra emissions of N and fossil CO_2. Using slurry on cash crops instead of fertiliser in cash crops creates more emissions of N (ammonia, nitrous oxide and nitrate), contributing to several environmental impact categories. It does not seem satisfactory to leave this as an extra burden on the cash crops.

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

Inventory for ‘1 kg slurry-N from pig farm’	
Artificial fertiliser, g N	600
Ammonia emitted, g N	69
Nitrous oxide emitted, g N	21
Nitrate emitted, g N	310
Sum, g N	1000
Characterized results for ‘1 kg slurry-N from pig farm’	
Global warming potential, g CO_2-eq.	578
Eutrophication potential, g NO_3-eq.	1750
Acidification potential, g SO_2-eq.	133

The method presented predicates that these emissions should burden the livestock products, but only after a proper systems expansion model has been established. Above it was presented how this may be done relatively easily. Due to the detailed Danish requirements for a proportion of fertiliser N to be replaced by slurry N there was a transparent reference for calculating avoided CO_2 and N emissions from saved fertiliser. In countries where this is not the case there is a need to develop an approach building on representative data regarding the degree of fertiliser replacement from slurry in the farming systems in question.

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

In the example described above, the system is expanded and the avoided production of artificial fertiliser included, and thus the approach completely follows the principles of consequential modelling. However, this system expansion, where the avoided productions are included, is not only used by LCA-researchers devoted to the use of consequential LCA (e.g., Weidema (2003); Schmidt (2007); Thrane (2006; 2005); Halberg et al. (submitted), but also by other LCA-researchers (Williams et al., 2006; Lopez-Ridaura et al., 2007), simply because it is obvious that slurry is a valuable fertiliser, which makes it possible for the farmer to save some artificial fertiliser. Thus slurry reduces the environmental impacts arising from production and distribution of artificial fertiliser.

Example 2: How data collection can be diminished due the consequential approach
In chapter 2 I described how the consequential approach seeks to model the technology (or process) actually affected by a change in demand. An example with electricity was given, where only the electricity sources affected by a change in demand are included in the modelling, whereas in the attributional approach the electricity consumption is often modelled as an average of all electricity sources within the region. In the example with electricity it is clear that the data collection for the consequential approach is easier because only the 'marginal' electricity source (for example coal-based electricity) is relevant. Similar to data collection for electricity, the data collection for feed mixtures is easier when using the consequential approach, because only the 'marginal' feed ingredients are relevant. Although it is more complicated compared to the electricity example. In the electricity example one kWh is one kWh irrespective of whether the primary energy source is coal or wind. A typical feed mixture for pigs, on the other hand, contains various ingredients (e.g., barley, soybean meal, rape seed meal, rape seed expeller, palm kernel expeller, soybean oil), and all these ingredients differ in their protein and energy content. In attributional modelling it would be necessary to provide LCA-data on each of these ingredients (e.g., Thomassen et al., 2007; Cederberg & Flysjö, 2004), while in the consequential modelling only LCA-data on the ingredients actually affected by a change in demand are relevant. The marginal protein source is soybean meal, meaning that an increased demand for pig feed (and thereby protein) will affect the soybean meal production and not, for example, the rape seed meal production, because soybean meal is the fastest expanding on the market (Paper 2). Demand for soybean meal has increased faster than total meat production, implying a net increase in the use of soybean meal per unit of meat produced (Steinfeld et al, 2006, p. 43).

The examples show that data collection can in some cases (e.g., electricity and feed ingredients) be reduced if consequential modelling is used instead of attributional modelling, although Curran et al. (2005) and Heijungs & Guinée (2007) state that the consequential modelling requires more data. In some cases they are right. For example in the LCA of soybean meal (Paper 2), where the consequential approach required more data, because the increased demand for soybean meal resulted in avoided production of palm oil. Then suddenly LCA data on palm oil production was required to perform an LCA on soybean meal. So whether one approach requires more data than the other depends on the defined system in the respective LCA, and no fixed rule can be established.

Allocation is one of the most sensitive issues in LCA methodology (Guinée, 2002). In consequential LCA, allocation is 'systematically' avoided and this is obviously and advantage because it increases the quality of an LCA. An example is the allocation of environmental impacts from the co-production of milk and beef. Should the allocation factor be related to the mass, the protein content, the energy content or the prices of the two products? I think the use of allocation factors is an arbitrary solution, which creates arbitrary results. In many cases the choice of allocation factor affects the result significantly. For example, in Paper 2 (table 5) where allocations based on mass and economy were performed. The global warming potential per kg soybean meal increased by 24% when mass allocation was used instead of economy.

An argument against the consequential approach is that the degree of uncertainty in identification of marginal products is too large (Heijungs & Guinée, 2007). To some extent it is right that identification of the marginal technology is not always straightforward. In Denmark there has, for example, been an on-going discussion on whether ‘marginal’ electricity is based on natural gas or coal. Is the marginal electricity technology the one that will be affected if a change in demand appears ‘tomorrow night’, or is it the one that will be affected if a change in demand appears for a number of years? A challenge for the researchers working with consequential modelling is obviously to make it more transparent how the marginal products (or technologies) can be identified, as this is often difficult for outsiders to understand. But it should be emphasized that although the identification of marginal products (or technologies) is uncertain, it is still much more preferable to using arbitrary allocation factors as is done in attributional LCA.

The attributional and consequential approaches are fundamentally different, not only seen in relation to allocation and system expansion. In the attributional approach the aim is to quantify the environmental impacts from the average products (e.g., average pig, average electricity), and it seeks to partition the environmental burden between the products coming from the same process (e.g., beef and milk). In contrast, the consequential approach seeks to estimate the environmental impact of products produced in the future. For example, in this thesis it is concluded that an increase in Danish pig production will increase not only the emissions of pollutants, but will also affect the soybean and oil palm production.

4.3 Quantification of emissions

In LCAs of agricultural products several pollutants are quantified and considerable quantities are emitted from biological systems (e.g., slurry, fields). The uncertainties in the quantification of some of the pollutants are high. This is especially the case for nitrous oxide, phosphate and CO_2 emitted from land-use changes. Issues regarding the quantification of these three pollutants will be briefly discussed in the following.

Nitrous oxide

In this study the guidelines from the Intergovernmental Panel of Climate Change (IPCC, 2000) were used to quantify the nitrous oxide emissions from the slurry and fields. In the LCA of pork it was shown that 72% of the greenhouse gases (measured in CO_2-eq.) emitted were nitrous oxides (figure 7), whereof most was emitted from the denitrification of leached nitrate or from artificial fertiliser-N applied to the fields. In the LCA of soybean meal (Paper 2) nitrous oxide was also the largest contributor to global warming potential. Consequently, nitrous oxide is important. Unfortunately, a closer look at the nitrous oxide emission factors in the guidelines from IPCC (2000), revealed that the emission factor (part of N lost as nitrous oxide) is 1.25% for both crop residues, artificial fertiliser-N applied to field, slurry-N applied to field and N fixation by crops, and that the uncertainty ranges for the many emission factors are in the interval -50% to 100%. This indicates that the method for calculating nitrous oxide losses from agricultural systems is simplified and highly uncertain, and in future LCAs of agricultural products it would be worthwhile to analyse if a more detailed (and site-specific) methodology exists on the quantification of nitrous oxide emissions from fields and slurry.

Crutzen et al. (2007) argue that the emission factors from IPCC on nitrous oxide losses from artificial fertiliser-N, slurry-N and crop residues were significantly underestimated and should be more than twice the amounts stated. However, if these higher emission factors suggested by Crutzen et al. (2007) were applied to the LCA of soybean meal (Paper 2) and LCA of pork (Paper 3), the conclusions regarding environmental hotspots would not be changed but only strengthened, because nitrous oxide emissions had already been identified as environmental hotspots in relation to global warming potential.

LCA is increasingly used in environmental assessment of agricultural production, and in order not to reach wrong conclusions, it is crucial that the methodology for quantifying nitrous oxide emissions from agricultural systems is developed and improved.

Phosphate

In this study the amount of phosphate leached was quantified calculating the P surplus and then assuming that phosphate leaching was directly proportional to the P surplus (figure 5). This means that farms with a high P surplus also had high phosphate leaching levels as shown in Paper 1 (tables 6 and 7). The assumption regarding direct proportionality between P surplus and P leaching is simple, because livestock farms, in general, have higher P surpluses (Paper 1) and therefore the P binding capacity of the soils will be used up on the livestock farms before the arable farms. As shown in figure 3, the livestock is centralised in specific regions, which means that the high P surpluses are also centralised in specific regions. Additionally, the areas in Western Jutland are also sandier, and sandy soils, in general, have a lower P binding capacity (Dalgaard & Rubæk, 2005). Therefore, there are reasons to believe that there is a higher risk of phosphate leaching from the livestock intensive areas, and especially those in Western Jutland. The soils in these areas have had P applied in excess for many years, thus leading to higher risk of phosphate leaching because the phosphate-binding capacity has been used up. Therefore it would be preferable if phosphate emissions could be quantified in a more site-specific context, which could take into account that P surpluses and phosphate leaching are not directly proportional. This would presumable lead to a higher eutrophication potential per kg pork. More phosphate would be leached on the pig farm, but maybe less would be leached from the fields where the feed grain is cultivated.

Phosphate only contributed 3% to the eutrophication potential (section 2.5 and Paper 3) and not to any other of the environmental impact categories. This could wrongly lead to the conclusion that phosphate leaching from the fields is not an environmental problem, and initiatives to reduce the phosphate leaching from pig farms are unnecessary. But phosphate leaching is an environmental problem in Denmark (Poulsen & Rubæk, 2005; Kronvang et al., 2001; Anonymous, 2004), and a reduction of phosphate leaching is needed. As described in chapter 1, P is the limiting factor for algal bloom in most Danish lakes (Kronvang et al., 2001). That phosphate emissions are overlooked in an LCA of pork is a consequence of the characterization, in which nitrate (and ammonia) becomes dominating. This highlights that LCA-based environmental indicators should not stand alone but be supplemented with area-based indicators, as for example P surplus or phosphate leached per hectare. Moreover, it would improve the quality of LCAs on agricultural products if site-specific characterization factors could be developed for phosphate.

CO_2 from land-use changes

Paper 2 described how the soybean area in Argentina is rapidly expanding at the expense of natural habitats. When natural areas (e.g., forest, savannah) are converted to soybean fields, organic carbon from above ground (e.g., stems) or from the soil will be degraded and CO_2 emitted. Logically, an increase in demand for Danish pork will therefore result in increased CO_2 emission, because of land-use changes in Argentina. Nevertheless, the emissions from land changes are not included in the LCAs of soybean meal (Paper 2) or pork (Paper 3). But in Paper 2, it was roughly estimated that the global warming potential per kg soybean meal would increase dramatically from 0.7 to 5.7 kg CO_2-eq. per kg soybean meal if the carbon released due to land-use changes was included. This changes the picture of the environmental hotspots in the product chains of soybean meal and pork completely. On the other hand, in the product chain of pork there will presumably also be parts where more organic carbon is incorporated into the soil. For example, if more pigs are produced, then more slurry is excreted, and if this slurry is used at cash crop farms instead of artificial fertiliser-N, there might be an increase in soil carbon status at the cash crop farm. So if CO_2 emissions from soil should be included, it should be performed consistently, and not only in one single segment of the product chain. Other large methodological gaps still exist (discount period, driving forces of deforestation, estimates of changes in above and below-ground carbon content as discussed in Paper 2) before it can sensibly be included in an LCA of livestock products. However, there is no doubt that land-use changes contribute significantly to global greenhouse gas emissions, and that they may be one of the largest problems of the world's fast growing livestock sector. Steinfeld et al. (2006) estimated that livestock related land-use changes may emit 2.4 billion tonnes of CO_2 per year. This value exceeds both the annual emission of nitrous oxides (2.2 billion tonnes CO_2-eq.) and methane (2.2 billion tonnes CO_2-eq.) from livestock activities. This indicates that CO_2 emitted as a consequence of land-use changes is a large environmental hotspot, and that one of the next improvements in the methodology of LCA of agricultural products should be to include these aspects consistently.

5 Suggestions for improved environmental regulation of the pig sector

In chapter 1 it was explained how the 'Nutrient farm account' system limits the use of slurry-N and artificial fertiliser-N in the agricultural sector. The 'Nutrient farm account' system encourages the farmers to treat their slurry with care in order not to loose its fertiliser value. If – for example – a large part of the N from the slurry is vaporised (e.g. because of an uncovered slurry tank) before it is applied to the fields, there will be less N available for the plants and the yields will be lower. Accordingly, it is profitable for a farmer to minimize the N losses. This is how it is supposed to work, but as for all other managing directors, some farmers are better managers than others, also when it comes to nutrient management.

In this thesis it was shown how N, in its different forms (nitrate, nitrous oxide and ammonia) is a major polluter in the product chain of pork. So even though the 'Nutrient account system' was launched with the intention of reducing eutrophication of the aquatic environment, it will hopefully lead to reduced acidification and global warming, because a more efficient use of N might also reduce the ammonia and nitrous oxide emissions.

Unfortunately, a similar 'Nutrient farm account' system is not used for the regulation of P. If the farmers were only allowed to apply a certain amount of P per hectare, the phosphate leaching would probably be reduced. Another option would be to apply an upper limit on the farmers' P-surplus per hectare. This would encourage farmers to use the P more efficiently and to limit the amount of P in feed imported to the farm.

Obviously, the impact of P on the aquatic environment is problematic (Kronvang et al., 2001; Poulsen & Rubæk, 2005), and might be a big barrier when Denmark has to fulfil the Water Framework Directive's requirements on the ecological quality in the aquatic environment. However, in the last year several research projects on phosphate pollution from the agricultural sector have been started. For example, the effect of farm management (e.g., soil preparation, liming) on the amount of phosphate leached is analyzed. Hopefully, these projects will provide knowledge of the most important factors leading to phosphate leaching, and they will facilitate the identification of areas that are most at risk of phosphate leaching.

In 2006 it was possible for farmers to get an up to 40% subsidy from rural development funds if they invested in a slurry separation plant (Frandsen, 2007), and many of the slurry separation plants were built in 2006 (Birkmose, 2007). Why rural development means were used to subsidise the establishment of slurry separation plants is not clear. But as highlighted in chapter 3 and Paper 5, there are some environmental problems associated with slurry separation, e.g., low separation efficiencies, N losses from the fibrous fraction, and it can be debated to what extent it is rational that public funds are used for the establishment of technologies, that have unsolved environmental problems and the effect of which are unknown. A thorough environmental evaluation ought to be performed both at farm level and at national level, to ensure that the subsidies do not support activities that negatively impact the environment. Moreover, it must be ensured that the farms separate efficiently, so sufficient P is exported from the farm. On a national level it must evaluated whether the establishment of slurry plants results in increasingly centralisation of the pigs in specific areas. If a pig farmer wishes to expand the pig herd and does not have sufficient farmland, the establishment of a slurry plant will permit the enlargement of the pig herd. Accordingly, the slurry separation plant is used as lever to obtain more pigs in an already pig crowded area. But is that a problem if the fibrous fraction is exported from the pig intensive area to the cash crop farms? Yes, for two reasons: i) the P load per unit area will not necessarily be decreased, if the pig herd is expanding and the P separation efficiency in the slurry separation plant is low, and ii) the ammonia emissions form the housing and storage will be increased if the pig herd is expanded, and slurry separation does not solve this problem.

In conclusion, if the use of slurry separation is not environmentally evaluated at farm level and at national level, there is a danger that the technology will be used as a means of expanding the pig herd in areas that already have a high livestock density, and thereby causing extra ammonia and phosphate loads in these areas.

In the LCA of pork (figure 6) it was shown that greenhouse gases were emitted not only from the pig farm, but also from other parts of the production as for example production, processing and transport of feed and artificial fertiliser. In Paper 2 it was estimated that if the greenhouse gas emission from the production and transport of the soybean meal consumed yearly by Danish livestock was included in the National Greenhouse Gas Inventory, the ‘Danish’ greenhouse gas emissions would be 12% higher. If the CO_2 emitted due to land-use changes caused by the soybean expansion in Argentina was also included, it would be much higher. In conclusion, the Danish pig production causes large emissions of greenhouse gases, which are not included in the National Greenhouse Gas Inventory. But should these greenhouse gas emissions from the southern hemisphere be ascribed to the Danish National Greenhouse Gas Inventory? The answer is no, because the difficulties in establishing the National Greenhouse Gas Inventories, which include the emissions from other countries, would be tremendous, and presumably not solve the global warming problem. However, it is crucial that all countries work for greenhouse gas reductions at a global level. A global problem such as greenhouse gas emissions must find a global solution, and this requires holistic approaches, exemplified by life cycle thinking. Take biofuel production: Before starting large scale biofuel production (based on, for example, maize or grain) in Europe with the aim of reducing fossil CO_2 emissions, it must be analysed to what extent an increased European energy crop production will increase greenhouse gas emissions from other continents. Will energy crops be produced at the expense of feed crops, thus enhancing the import of feed crops from other continents, which again will intensify their agricultural production at the expense of natural habitats and finally resulting in increased greenhouse gas emissions? Action must be taken in order to prevent that type of pollution swapping. In the longer term it will be necessary to develop international agreements or legislation that can prevent a country can decrease its greenhouse gas emissions by reducing production (of, for example, gasoline) but increasing import (of, for example, animal feed).

6 Conclusion

The most polluting parts of the Danish pork production chain were the farms where the weaners (size: 30 kg) and the fattening pigs (size: 100 kg) were produced. These were the two major contributors to both global warming, eutrophication and acidification potentials. However, the two major contributors to photochemical smog potential were the feed purchased to the farm and production and distribution of electricity and heating oil used on the farm. The contribution from transport, processing of the meat and buildings were very limited.

The most contributing substances from the product chain of pork were nitrous oxide (53%) and methane (10%) for global warming potential, nitrate (63%) and ammonia (30%) for eutrophication potential, ammonia (84%) for acidification potential and volatile organic carbons (86%) for photochemical smog potential. Because nitrate, nitrous oxide and ammonia all contain N, a more efficient use of N at the pig farms and in the feed production will obviously improve the environmental profile of pork.

Addition of the feed digestibility-improving enzyme xylanase to the pig feed was shown to improve the environmental profile of pork. The global warming potential per kg pork could be reduced by approximately 5% if xylanase was added to the feed, whereas the reduction in eutrophication potential was limited.

Slurry separation is a technology where slurry is separated into a liquid and a fibrous fraction. It was shown that if the fibrous fraction (which has a high P and low water content) was exported out from the pig farm the amount of slurry transported and P applied at the fields on the pig farm could be reduced by respectively 37% and 82%, compared to a situation where un-separated slurry was exported. However, these environmental improvements required that the slurry separation plants were separating efficiently, and this was not the case at the two private pig farms where data were collected. Substantial reductions (approx. 16%) in greenhouse gas emissions per kg pork could be obtained if the slurry was anaerobically digested and the biogas was used for heat and power production. On the other hand the anaerobic digestion did not change the nutrient contents in the slurry, and had therefore not the same potential of reducing the P loads on the pig farms as slurry separation had.

LCA is a valuable tool for environmental assessment of livestock products, especially when used in combination with area-based indicators. But there still is a need for developing the methodology used for quantification of nitrous oxide and phosphate emissions. Also quantification of CO_2 emissions related to land-used changes is a great challenge.

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1

Modelling representative and coherent Danish farm types based on farm accountancy data for use in environmental assessments

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Modelling representative and coherent Danish farm types based on farm accountancy data for use in environmental assessments

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Abstract

There is a need for valid and representative data on the production, resource use and emissions from different farm types in Denmark for public regulation and assessment. The data should be usable for both area-based environmental assessment (e.g. nitrate leaching per ha) and product-oriented environmental assessment (e.g. greenhouse gas emissions per kg pork). The objective of this study was to establish a national agricultural model for estimating data on resource use, production and environmentally important emissions for a set of representative farm types.

Every year a sample of farm accounts is established in order to report Danish agro-economical data to the 'Farm Accountancy Data Network' (FADN), and to produce 'The annual Danish account statistics for agriculture'. The farm accounts are selected and weighted to be representative for the Danish agricultural sector, and similar samples of farm accounts are collected in most of the European countries. Based on a sample of 2138 farm accounts from year 1999 a national agricultural model, consisting of 31 farm types, was constructed. The farm accounts were grouped according to the major soil types, the number of working hours, the most important enterprise (dairy, pig, different cash crops), livestock density, etc. For each group the farm account data on the average resource use, products sold, land use and herd structure were used to establish a farm type with coherency between livestock production, feed use, land use, yields, imported feed, homegrown feed, manure production, fertilizer use and crop production. The set of farm types was scaled up to national level thus representing the whole Danish agricultural sector and the resulting production, resource use and land use was checked against the national statistics. Nutrient balance methodology and state-of-the-art emission models and factors were used to establish the emissions of nitrate, phosphate, ammonia, nitrous oxide, methane and fossil carbon dioxide from each farm type. In this paper data on resource uses and emissions from selected farm types are presented and it is demonstrated that this approach can lead to an agro-environmental inventory, which is consistent with national level estimates and still has the advantage of being disaggregated to specific farm types. Conventional dairy farm types in general emitted more nitrate but less phosphate compared with pig farm types. The methane emission was higher from dairy farm types compared with all other farm types. In general the conventional dairy farms emitted more nitrate, ammonia, and nitrous oxide, compared with organic dairy farms.

Keywords: Agriculture; Environmental assessment; Environmental impact; Emissions inventory; Nutrient balances

1. Introduction

Agricultural production has an impact on the environment on a local scale (e.g. nitrate leaching to fens) and on a global scale (e.g. greenhouse gas emissions to the atmosphere). In order to identify the most polluting sources of the agricultural production it is crucial to use well-defined environmental indicators and valid data to describe resource use and emissions from different farm types.

Environmental indicators developed for agricultural purposes have recently been reviewed by Halberg et al. (2005) and Payraudeau and van der Werf (2005). Halberg et al. (2005) distinguish between area-based indicators

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(e.g. nitrate leaching per ha) and product-based indicators (e.g. greenhouse gas emissions per kg product) and conclude that both types of indicators are needed in order to comprehensively characterise environmental impacts from food production.

Area-based indicators are useful for evaluating farms emissions of nutrients such as nitrate, ammonia and phosphate that all have an effect on the local environment. In several studies (Jarvis and Aarts, 2000; Haas et al., 2001; Holbeck and Hvid, 2004; Jarvis and Menzi, 2004; Kristensen et al., 2005a; Nielsen and Kristensen, 2005) area-based indicators have been used to compare nutrient surpluses from different farm types. As data-collection from farms is time-consuming, these studies are based on a limited number of farms and are not statistically representative for the agricultural sector. The lack of representative data for environmental indicators and assessment may be misleading because results such as comparison between farm types may be highly influenced by individual farm performances.

Product-based indicators are useful for evaluating the impact of food productions on the global environment (e.g. climate change) and have the advantage that in addition to emissions from the farms, emissions related to the production of inputs (e.g. soybean, artificial fertilizer) and outputs (e.g. manure exported to other farms) are also included. Life-cycle thinking is the basic idea behind the product-based indicators.

Life-cycle thinking is one of five key principles in the European Union's Integrated Product Policy (IPP) (European Commision, 2003) and is also supported by the United Nations Environmental Programme (UNEP, 2004). In Life-cycle thinking the cradle-to-grave approach for a product is adopted to reduce its cumulative environmental impacts (European Commision, 2003). The most developed tool for life-cycle thinking is life cycle assessment (LCA), which is a method of evaluating a product's resource use and environmental impact throughout its life-cycle. LCA has been used for environmental assessment of milk (Cederberg and Mattsson, 2000; Haas et al., 2000; Thomassen and de Boer, 2005), pork (Cederberg and Flysjö, 2004; Eriksson et al., 2004; Basset-Mens and van der Werf, 2005), grains (Weidema et al., 1996) and other agricultural products, but most of the existing LCAs are based on data from only one or a few farms. However, there is considerable variation in the resource use and emissions between farms of the same main enterprise (Halberg, 1999; Haas et al., 2000; Weidema et al., 2002; Thomassen and de Boer, 2005) and it is therefore unsatisfactory to base evaluation and comparison of agricultural products on case studies.

In order to produce representative area-based and product-based environmental indicators, there is a need for representative and valid farm data that describes resource use and emissions from typical farms.

Poppe and Meeusen (2000) and Halberg et al. (2000) proposed basing environmental assessments on representative farm accounts such as those collected for The Farm Accountancy Data Network (FADN). The aim of FADN is to gather accountancy data from farms for income determination and business analysis of agricultural holdings. The annual sample of FADN covers approximately 80,000 holdings in Europe, that represent about 5,000,000 farms, thus covering approximately 90% of the agricultural area and more than 90% of the total agricultural production of the European Union (FADN, 2006). For each farm sampled, the data relates to variables such as livestock, agricultural area, crop yields, etc. FADN is an instrument for evaluating the income of agricultural holdings and the impacts of the Common Agricultural Policy. We found that FADN could also be used as the data source for performing area-based and product-based environmental assessments.

The objective of this study was to establish a national agricultural model to estimate resource use, production and environmentally important emissions based on a set of representative farm types.

The national agricultural model should be able to deliver data for both area-based environmental assessments (e.g. nitrate leaching per ha, methane emissions per ha) and product-based environmental assessments (e.g. global warming potential per kg pork). This paper gives results in terms of representative farm types, their resource uses and emissions per ha. Per hectare results are given in kg N and P farm gate balances, nitrate–N, ammonia–N, nitrous oxide–N, phosphate–P, methane and fossil carbon dioxide.

2. Methods

The Danish agricultural sector was divided into 31 representative farm types. For each farm type data describing farm type characteristics (e.g. agricultural area, crop yields) were averaged over a number of farm accounts from private farms. Based on this, resource use (e.g. import of soybean meal, diesel, artificial fertilizer) and products sold (e.g. pork, cereals) from the farm types were modelled. Emissions (e.g. methane, nitrate, ammonia) were also calculated from the modelling of nutrient cycling and flows of energy and materials. Point of departure for modelling of the farm types was a set of representative farm accounts as explained below.

2.1. Farm account statistics

Danish farmers are obliged to keep records of purchases and sales for tax purposes and the annual accounts are made with professional help. Every year a sample of these farm accounts are collected by Food and Resource Economic Institute in order to fulfil Denmark's obligation to supply FADN with farm data, and to produce 'The annual Danish account statistics for agriculture' (Møllenberg, 2001; Larsen, 2003).

In the year 1999 the sample contained 2138 farm accounts with detailed data describing the farms' agricul-

Table 1
Criteria used for partitioning of farm accounts among farm types (sandy loam soil)

Farm type	1	2	3	4	5	6	7	8	9	10	11	12	13	30
Name	Part-time	Sugar beets	Grass seeds	Milk	Milk	Milk	Organic milk	Pig	Pig	Pig	Cash crops	Residual	Horticulture	Organic plant
Type of criteria														
Working hours per year[a]	<832	>832	>832	>832	>832	>832	>832	>832	>832	>832	>832	>832	>832	>832
Conventional (C)/organic (O)	C[b]	C[b]	C[b]	C[b]	C[b]	C[b]	O[c]	C[b]	C[b]	C[b]	C[b]	C[b]	C[b]	O[c]
Specialization	Non	Sugar beets[d]	Grass seeds[e]	Milk[f]	Milk[f]	Milk[f]	Milk[f]	Pig[g]	Pig[g]	Pig[g]	Cash crops[h]	Non[h]	Horticulture[i]	Organic plant[j]
Livestock density (LU ha^{-1})[k]		<1.4	<1.4	<1.4	1.4–2.3	>2.3		<1.4	1.4–1.7	>1.7	<0.5	>0.5		
Distribution of farm accounts														
Sample[l]	67	88	63	23	32	14	24	50	27	98	53	38	185	30
Population[m]	5663	2009	1616	432	849	267	115	1322	424	1437	1983	1219	1133	289
Percent of total production in Denmark														
Milk	0%	2%	0%	4%	7%	3%	1%	0%	0%	0%	0%	2%	0%	0%
Fattening pigs	0%	4%	3%	0%	0%	0%	0%	4%	4%	21%	0%	3%	0%	0%

[a] Part-time holdings: less than 832 working hours per year.
[b] C: holdings that did not receive organic subsidies.
[c] O: holdings that did receive organic subsidies.
[d] Holdings where at least 10% of the area was cultivated with sugar beets.
[e] Holdings where at least 10% of the area was cultivated with grass seeds.
[f] Holdings with dairy cattle. Maximum 10% of gross margin came from pigs.
[g] Holdings with pigs. Minimum 10% of gross margin came from pigs. Maximum 10% of gross margin came from cattle.
[h] Residual holdings not applying to previous criteria.
[i] Horticultural accounts were marked specific, and could therefore be selected for this farm type.
[j] Residual holdings that did receive organic subsidies, but had no dairy cattle.
[k] One LU equals to: 1 milking cow, 3 sows with piglets (to 25 kg) or 24 fattening pigs (30–110 kg).
[l] Number of farm accounts that fulfilled the criteria of the respective farm type.
[m] Number of Danish farms that the farm type represented.

tural area (e.g. number of hectares of spring barley and rape seed), crop yields (e.g. kg cereals, rape seed, potatoes and grass seeds per ha), livestock products sold (e.g. kg milk, meat, live animals), livestock density, electricity use, etc. The farm accounts were weighted and selected to be representative for the entire population of 50,487 Danish farms. Firstly, the farm accounts were divided into two groups according to their main soil type. The sandy loam soil group was composed of farm accounts where the majority of the agricultural area had soil containing more than 10% clay, whereas the sandy soil group contained the rest. The farm accounts from farms in the sandy loam soil group were, subsequently, divided into 14 groups using the criteria presented in Table 1. For each farm type on sandy loam soil a set of criteria regarding the number of working hours per year, organic subsidies, specialization (e.g. sugar beets, milk, pigs) and livestock density (livestock units ha^{-1}) was defined. The criteria regarding number of working hours and specialization were used in order to separate small mixed farms from large specialized farms. Thereby the modelling was facilitated and the farm types were reflecting the structure of the Danish agricultural sector, which is moving towards larger and more specialized farms. The criterion of livestock density was used to separate livestock farms from cash crop farms, and to secure that the modelling of manure exchange between farm types could be performed in accordance with the public regulation of manure and fertilizer use in Denmark, which partly is based on livestock density (Plantedirektoratet, 1998). Organic farms and conventional farm accounts were separated to secure that artificial fertilizer was not purchased by organic farms.

Secondly, all farm accounts belonging to the sandy loam soil group were tested against the criteria of farm type 1. The number of working hours should be less than 832 $year^{-1}$ and the farm should not receive organic subsidies. The criteria were fulfilled by 67 farm accounts, representing a population of 5663 Danish part-time farms. The remaining farm accounts were tested against the criteria of farm type 2, and 88 farm accounts matched the four criteria, namely more than 832 working hours $year^{-1}$, not receiving organic subsidies, at least 10% of the area cultivated with sugar beets and maximum 1.4 livestock units ha^{-1}. This procedure was followed to divide all the farm accounts for sandy loam soil farms in a sequential procedure using the criteria under each predefined type as shown from left to right in Table 1. The same was done to farm accounts in the sandy soil group (Table 2). For further details, see Larsen (2003).

Farm accounts with more than 10% of gross margin from poultry were not divided according to soil type, but were partitioned into farm types 27, 28 and 29 (not presented in Tables 1 and 2).

After the partitioning of the farm accounts into the 31 farm types, the data in the farm accounts belonging to the same farm type were averaged, and thus each farm type was represented as one averaged farm account, containing data describing the agricultural area, crop yields, livestock production, purchased inputs, etc. The existing system of sampling did unfortunately not permit calculation of variance on data from the farm types. However, on a European level, the Commission has since 1965 used the farm account data to determine differences between farm types in their productivity and economic competitiveness (FADN, 2006), and therefore we also found it suitable for comparison of environmental aspects.

For each farm type a detailed model was then developed, based partly directly on the averaged farm accounts, and partly on general knowledge as explained in the following.

2.2. Modelling coherent and representative farm types

The data in the farm accounts contained information on the agricultural area, crop yields, livestock products sold, livestock density, electricity use, etc. of each farm type. This information was thus used to establish the general crop–livestock interaction (e.g. how much homegrown barley was used as feed on the farm, how much manure was used for fertilization), and the level of production within each farm type. Because the use of external inputs like purchased feed and fertilizer was only available in the monetary units Danish Kroner (DKK) in the accounts, the feed and fertilizer use in kg nutrients was modelled using standards. The use of electricity and chemicals in DKK and the production of fattening pigs and milk were not modelled but were averaged data from the farm accounts.

Due to the public regulation of manure and fertilizer use in Denmark, representative average values for feed efficiency in livestock production (e.g. feed use per kg live weight pig) and the production of nitrogen (N) and phosphorus (P) in manure by livestock types are readily available. For example a fattening pig (30–100 kg) by standard excretes 5.1 kg N and 0.72 kg P and has an N efficiency of 0.38 (Poulsen et al. (2001), an update from Poulsen and Kristensen (1998)). N efficiency is defined as the N produced in the carcass divided by the N intake in feed, and was lower for sows and piglets in comparison with fattening pigs. The N demand and N excretion of a milking cow were also calculated according to Poulsen et al. (2001), an update from Poulsen and Kristensen (1998), but as they depend on the milk yield per cow and on the percentage of Jersey stock, they were calculated for each the farm type using the farm accounts data. N demand and N excretion were 150–176 and 114–133 kg N cow^{-1} $year^{-1}$, respectively. N-efficiency of 0.25 and 0.24 for Jersey and dual-purpose breed cows, respectively, were used (Poulsen et al. (2001), an update from Poulsen and Kristensen (1998)).

The amount of homegrown feed was estimated by multiplying farm account data on area by yields per hectare of feed crops, which were obtained from private pilot farms (Kristensen et al., 2005a). Then the purchase of external feeds was modelled as the difference between the livestock's protein and energy needs and the input from homegrown (Halberg et al., 2000).

Table 2
Criteria used for partitioning of farm accounts among farm types (sandy soil)

Farm type	14	15	16	17	18	19	20	21	22	23	24	25	26	31
Name	Part-time	Potatoes	Milk	Milk	Milk	Organic milk	Pig	Pig	Pig	Suckler cows	Cash crops	Residual	Horticulture	Organic plant
Type of criteria														
Working hours per year[a]	<832	>832	>832	>832	>832	>832	>832	>832	>832	>832	>832	>832	>832	>832
Conventional (C)/organic (O)	C[b]	C[b]	C[b]	C[b]	C[b]	O[c]	C[b]	C[b]	C[b]	C[b]	C[b]	C[b]	C[b]	O[c]
Specialization	Non[d]	Potatoes[e]	Milk[f]	Milk[f]	Milk[f]	Milk[f]	Pigs[g]	Pigs[g]	Pigs[g]	Suckler cows[h]	Cash crops[i]	Non[i]	Horticulture[j]	Organic plant[k]
Livestock density (LU ha^{-1})[l]		<1.4	<1.4	1.4–2.3	>2.3		<1.4	1.4–1.7	>1.7		<0.5	>0.5		
Distribution of farm accounts														
Sample[m]	59	62	83	182	16	125	99	38	164	103	52	91	100	107
Population[n]	5043	1184	1912	4004	330	695	2319	600	2347	6309	2103	2229	644	1084
Percent of total production in Denmark														
Milk	0%	2%	15%	43%	4%	9%	0%	0%	0%	0%	0%	7%	0%	0%
Fattening pigs	0%	2%	0%	0%	0%	0%	13%	4%	30%	2%	0%	6%	0%	0%

[a] Part-time holdings: less than 832 working hours per year.
[b] C: holdings that did not receive organic subsidies.
[c] O: holdings that did receive organic subsidies.
[d] No suckler cows.
[e] Holdings where at least 10% of the area was cultivated with potatoes.
[f] Holdings with dairy cattle. Maximum 10% of gross margin came from pigs.
[g] Holdings with pigs. Minimum 10% of gross margin came from pigs. Maximum 10% of gross margin came from cattle.
[h] Holdings with suckler cows.
[i] Residual holdings not applying to previous criteria.
[j] Horticultural accounts were marked specific, and could therefore be selected for this farm type.
[k] Residual holdings that did receive organic subsidies, but had no dairy cattle.
[l] One LU equals to: 1 milking cow, 3 sows with piglets (to 25 kg) or 24 fattening pigs (30–110 kg).
[m] Number of farm accounts that fulfilled the criteria of the respective farm type.
[n] Number of Danish farms that the farm type represented.

Moreover, each farm has a fertilizer quota based on official crop N norms multiplied by the area with different crops. From this is deducted the plant-available N in the either home-produced or imported farm manure. For example, a cash crop farm on sandy loam soil with no manure production could import 117 kg N in artificial fertilizer ha^{-1} of spring barley if the field has carried a cereal crop the previous year (Plantedirektoratet, 1998). Thus, fertilizer use on the different farm types was calculated using these norms. As part of Danish compliance with the EU Nitrate Directive, the use of manure–N is limited, why some farms are obliged to export manure to cash crop farms. This was modelled as a transfer of manure from farm types with a high livestock density to other farm types, which then reduced the artificial fertilizer input accordingly. In 1999 the use of manure on pig farms was limited to 170 kg N ha^{-1}. Dairy farm types with a straw shortage were supplied with straw from farm types with a low livestock density. Because the model accounts for the entire land use and agricultural production on national level the consistency of exchange of manure and straw between farms could be checked.

Livestock farms are primarily situated in the western part of Denmark, where sandy soil predominates. This is reflected in the national agricultural model, where 80% of the milk and 57% of the fattening pigs were produced on sandy soil farm types (Table 2). It was assumed that transfer of manure and straw for bedding between the western and eastern parts of Denmark, which are separated by two straits, only occurred on a minor scale, and therefore the transfer of manure and straw between farm types was modelled to only occur between farm types belonging to the same soil group. Consequently, farm types on sandy soil interchanged more manure and straw than farm types on sandy loam soil.

In this way a coherent model of crop–livestock interactions was established for each farm type with a consistent relation between livestock production, use of homegrown versus imported feed and export of cash crops. There was also coherence between the farm types in terms of manure and straw. Moreover the use of N in artificial fertilizer and manure on all farm types was in harmony with Danish legislation.

Energy use for traction was modelled following Dalgaard et al. (2001), where each crop was assigned a number of field operations multiplied by diesel use per ha. It was assumed that the diesel consumption for passenger car driving was 5 l ha^{-1} $year^{-1}$, while the average distance from farm to field was 2 km. Electricity use was estimated directly from the data in the farm accounts using a standard price per kW h consumed, but corrected against national statistics.

2.3. Modelling farm nutrient balances and emissions

For each of the 31 farm types, N and P balances were established at farm gate level, herd level and field level following the procedures of Halberg et al. (1995) and Kristensen et al. (2005a). The N input to the farm types included: feed concentrates, straw, artificial fertilizer, manure, deposition, biological N fixation (BNF) and living animals. The BNF in pure legumes was calculated as a proportion of the yield in grain legume multiplied by the standard N content (Høgh-Jensen et al., 2004). In grass clover BNF was set at the average value obtained for approximately 100 private pilot farms during the period 1989–2003. BNF was estimated from the percentage of clover in the season based on 300 visual clover estimations per farm per year as described by Kristensen et al. (1995). The estimated BNF in organic grass/clover was 150 kg N ha^{-1} $year^{-1}$ and in conventional grass/clover fertilized with around 130 kg N ha^{-1} $year^{-1}$ as artificial fertilizer the estimated BNF was 100 kg N ha^{-1} $year^{-1}$. The N output from the farm types included: Meat, milk, crops and manure sold to other farm types.

The amounts of nitrate, ammonia, phosphate and greenhouse gasses (methane, nitrous oxide and fossil carbon dioxide) emitted from the farm types were determined on the basis of nutrients balances in combination with farm account data on agricultural area, livestock and internal flows.

Nitrate leaching was assumed to be equal to the N farm gate balance minus ammonia losses, denitrification (Kristensen et al., 2005a) and net change in soil N status. The ammonia emission from animal housing, manure storage and handling was calculated using standard values from Hutchings et al. (2001). These values are also presented by Kristensen et al. (2005a, see appendix). Denitrification was estimated using the method of Vinther and Hansen (2004), and the net change in soil N status was modelled using the dynamic soil model from Gyldenkærne et al. (2005), implemented in C-TOOL (Petersen et al., 2002).

The nitrous oxide emissions were calculated according to IPCC (2000), but using a country-specific accounting method for some of the crop residue N content (Møller et al., 2000).

According to 'Evaluation of the Action Plan for the Aquatic Environment II' (Action Plan for the Aquatic Environment, 2003) 1000 t of P leached in 2004, corresponding to 0.4 kg P ha^{-1}. In order to reach that level for the national agricultural model and to obtain proportionality between P farm gate balance and leaching it was assumed, that 2.9% of the P farm gate balance leached as phosphate.

The methane emission was calculated using standard IPCC methodology (IPCC, 2000). The methane emission from the cattle's enteric fermentation was calculated using data on dry matter intake from the farm models in combination with the Tier 2 method (IPCC, 2000). A methane conversion rate of 0.06 and energy content of 18.45 MJ kg^{-1} dry matter in feed was used. As the feed intake in the farm models varied with the milk yield per cow, the methane emission per dairy cow per year varied between farm types. The methane emission from manure management was calculated using the Tier 2 method, except for the methane conversion factor where the original standard of 0.10 (IPCC, 1997) was used instead of 0.39, as argued by Massé et al. (2003).

The emission of CO_2 from combusted fossil fuel was assumed to be 91 g CO_2 MJ^{-1} diesel and 94 g CO_2 MJ^{-1} heating oil (Nielsen et al., 2003).

Emissions and resource use relating to the construction and maintenance of buildings and machinery used on the farm were not included, and the use of medicine and pesticides was not considered. Emissions and resource use associated with the production of purchased resources (e.g. soybean meal, fertilizer) were not included in this study.

Estimates of uncertainty on the N farm gate surplus were calculated using a study by Kristensen (2005, a short translation of the Danish report by Hvid et al. (2004)), who found the uncertainties to be 8, 21, 13 and 18% (measured as coefficients of variation (CV)), respectively, for the farm types conventional and organic dairy, pig farms and non-livestock farms. These uncertainties were calculated on basis of standard deviations for each item in the N farm gate balance. The lowest CVs were on artificial fertilizer (5%) and milk (3%), whereas the highest were on BNF (25%) and on cash crops (20%). Those CVs were used to calculate the standard deviations for the actual items. Farm types with a large area of clover grass (e.g. organic dairy farm) and thus high N-input from BNF, also had high uncertainty on the N farm gate surplus. For further details on calculation of CVs and uncertainties, see Kristensen (2005). The uncertainties on the N farm gate balances were used in our study to indicate whether the differences in N farm gate balances between farm types were important. Due to the importance and variability analyses of the BNF estimate we also performed sensitivity analyses of changes in this parameter.

The emissions of N in the form of ammonia, nitrous oxide and nitrate were estimated in a coherent way, so that sum of the partial emissions and the net change in soil N status equalled the N farm gate balance. It was beyond the scope of this study to determine uncertainties on these items. But as explained above we have used international recognized methods for calculation of greenhouse gas emissions and national recognized methods for calculation ammonia emissions. In addition we checked the total estimated emissions against a separate national level inventory of emissions as explained in the following.

2.4. *Securing consistency with the national statistics*

To secure a consistency of the farm types with the national statistics, a three-step validity check was performed.

Firstly, a validity check of farm type production data against national statistics was performed. This was done by multiplying production data (e.g. number of milking cows, agricultural area, pigs produced) from each farm type with the number of farms the farm type represented (population in Tables 1 and 2), and then summarizing these multiplied data across all farm types and comparing the values with national statistics (Agricultural Statistics, 2000).

Secondly, the modelled resource use (e.g. soybean meal, diesel, artificial fertilizer N) of the farm types was compared with national statistics by similarly multiplying the resource use of each farm type with the population of the farm type, and then comparing these values with national statistics. The total use of artificial fertilizer N was underestimated. Therefore, figures for the farm types were adjusted using an overall factor on the input to all farm types. The model also underestimated the total use of diesel and heating oil, and the farm types were therefore adjusted accordingly.

Thirdly, the emissions of nitrous oxide and methane from farm types were compared with national statistics for emissions of greenhouse gasses (Gyldenkærne and Mikkelsen, 2004).

Area-based environmental indicators were calculated on the basis of the modelled farm types and presented as N and P farm gate balances and emissions of nitrate, phosphate, ammonia, methane, nitrous oxide and fossil CO_2 per ha.

3. Results

The results from the modelling of farm types and their consistency with national statistics are presented followed by results from the farm types in terms of emissions per ha.

3.1. *Establishment of farm types and their consistency with national statistics*

To secure the consistency of the national agricultural model based on representative farm types, production data and resource use across all 31 representative farm types were aggregated using the population of the respective farm types (Tables 1 and 2). The production data for pig and milk production and land use were in good agreement with the Danish National Statistics (Agricultural Statistics, 2000) as shown in Table 3. The farm types did not, however, account satisfactorily for the total use of artificial fertilizer N. The unexplained difference was corrected using an overall factor of 10% on the artificial fertilizer N input to all farm types. The total use of diesel and heating oil was underestimated by 18%, and the farm types were therefore adjusted accordingly. The underestimation of diesel use might be due to underestimation of passenger car driving or the distance from farm to field.

The aggregated emissions of nitrous oxide and methane across all farm types were 22,000 t N_2O and 160,000 t CH_4, and thereby the nitrous oxide emission was 9% higher and the methane emission 10% lower than the Danish National Statistics for emissions of greenhouse gasses (Andersen

Table 3
Aggregated production data and resource use across 31 representative farm types, scaled to national level and compared with the Danish National Statistics

	Farm types scaled to national level	Danish National Statistics[a]	Deviation from Danish National Statistics (%)
Production data			
Fattening pigs produced[b] (1000)	20639	20801	−1
Sows (yearly basis) (1000)	1083	1052	3
Milking cows (1000)	633	661	−4
Milk production (1000 t)	4624	4455	4
Agricultural area (1000 ha)	2585	2644	−2
Area with cereals (1000 ha)	1395	1448	−4
Area with roughage (1000 ha)	567	570	−1
Resource use			
Artificial fertilizer N (1000 t N)	226	252	−10
Soybean meal (1000 t N)	142	156	−9
Grain (1000 t)	6571	6728	−2
Diesel and heating oil (PJ)	13.0	15.8	−18

[a] Agricultural Statistics (2000).
[b] Live weight = 100 kg.

et al., 2001; Gyldenkærne and Mikkelsen, 2004). The difference in nitrous oxide emission was expected since we used more detailed information for crop residues than in the national nitrous oxide budget.

3.2. Selected farm types' characteristics and resource use

After correcting for national level consistency in terms of artificial fertilizer N, diesel and heating oil, the representative and coherent farm types showed the relationship between resource uses and emissions and specific volumes of livestock and cash crop productions. Detailed results are presented as an open database (Nielsen et al., 2003). Agricultural area, yields and resource use for selected farm types on sandy loam soil and sandy soil are presented in Tables 4 and 5, respectively. In Tables 4 and 5 the following parameters are modelled: Self-sufficiency in feed, grain for feed stock, soybean meal, manure, artificial fertilizer, heating oil and diesel. The rest of the parameters are farm account data.

On pig farms 71–79% of the area was cropped with grains compared with 14–48% on dairy farms. Organic dairy farms had the largest area (50–55%) with grass-clover, permanent grass and set-aside. The self-sufficiency in terms of feed (calculated on the basis of Scandinavian Feed Units) decreased with increasing livestock density and was in general higher for dairy farm types (64–85%) than for pig farm types (30–67%).

Table 4
Characteristics and resource use per year for selected farm types on sandy loam soil

Farm type Name	2 Sugar beets	3 Grass seeds	4 Milk	5 Milk	7 Organic milk	8 Pig	9 Pig	11 Cash crops
Livestock density (LU ha^{-1})	0.5	0.2	0.9	1.7	1.2	0.7	1.6	0.1
Population (number of farms)	2009	1616	432	849	115	1322	424	1983
Agricultural area (ha)	78	105	99	50	88	58	77	68
Grain (%)	60	57	48	32	24	77	79	74
Other cash crops (%)	30	35	6	3	3	14	11	15
Maize and whole crops (%)	1	0	17	29	23	0	1	0
Grass/clover in rotation (%)	1	0	14	24	33	1	1	2
Permanent grass, set aside (%)	8	8	15	12	17	7	8	9
Yields								
Wheat (kg ha^{-1})	8630	8030	6700	6830	5850	6950	7260	7510
Spring barley (kg ha^{-1})	6180	5940	4790	4970	3650	5370	5760	5360
Winter barley (kg ha^{-1})	6470	6080	5390	5370	–	6160	6000	5600
Rape seed (kg ha^{-1})	3040	3170	3040	2710	3000	3100	3170	2880
Self-sufficiency in feed (%)	72	75	83	64	74	67	36	91
Resource use								
Grain for feed stock (kg ha^{-1})	0	0	0	1294	1116	117	2214	0
Soybean meal (kg ha^{-1})	483	219	594	1402	250	763	1532	13
Manure (kg N ha^{-1})	6	7	6	0	22	7	0	16
N artificial fertilizer (kg N ha^{-1})	110	120	107	89	0	116	95	136
P artificial fertilizer (kg P ha^{-1})	10	12	10	11	0	9	5	15
K artificial fertilizer (kg K ha^{-1})	50	46	27	17	0	41	29	53
Electricity (kW h ha^{-1})	327	284	464	598	446	456	951	177
Heating oil (MJ ha^{-1})	440	229	5	19	2	1121	2334	334
Diesel (l ha^{-1})	140	120	147	166	124	140	148	178
Chemicals (DKK ha^{-1})	746	578	333	299	4	572	526	538

Table 5
Characteristics and resource use per year for selected farm types on sandy soil

Farm type	15	16	17	19	20	21	23	24
Name	Potatoes	Milk	Milk	Organic milk	Pig	Pig	Beef	Cash crops
Livestock density (LU ha^{-1})	0.4	1.1	1.8	1.4	0.8	1.5	0.7	0.0
Population (number of farms)	1184	1912	4004	695	2319	600	6309	2103
Agricultural area (ha)	94	81	65	102	76	79	31	76
Grain (%)	51	41	19	14	71	76	49	70
Other cash crops (%)	31	5	1	1	17	11	5	16
Maize and whole crops (%)	4	20	37	30	1	0	4	1
Grass/clover in rotation (%)	3	18	26	41	1	1	16	0
Permanent grass, set aside (%)	10	16	17	14	11	11	26	12
Yields								
Wheat (kg ha^{-1})	5930	6470	5490	4970	6620	6260	5620	6400
Spring barley (kg ha^{-1})	4600	4710	4550	3920	4920	4460	4290	4570
Winter barley (kg ha^{-1})	4790	5370	5580	2400	5430	5550	4420	5080
Rape seed (kg ha^{-1})	1960	2410	2200	1500	2640	2580	1840	1820
Self-sufficiency in feed (%)	79	85	66	71	57	30	95	97
Resource use								
Grain for feed stock (kg ha^{-1})	0	0	1411	1415	482	2619	0	0
Soybean meal (kg ha^{-1})	316	604	1185	345	843	1548	125	3
Manure (kg N ha^{-1})	16	8	0	20	7	0	11	16
N artificial fertilizer (kg N ha^{-1})	110	108	101	0	92	72	8	111
P artificial fertilizer (kg P ha^{-1})	10	11	14	0	9	2	7	14
K artificial fertilizer (kg K ha^{-1})	54	35	39	0	39	21	32	50
Electricity (kW h ha^{-1})	445	429	648	541	486	813	262	189
Heating oil (MJ ha^{-1})	241	7	11	5	1056	2813	56	3
Diesel (l ha^{-1})	128	143	164	122	122	132	110	107
Chemicals (DKK ha^{-1})	749	343	281	3	466	407	206	493

The grain yields of wheat, spring barley and winter barley, calculated as an area weighted average across all farm types, were 29% higher on farm types on sandy loam soil. This is presumably because of a higher soil fertility (Halberg and Kristensen, 1997). Organic dairy farms had the lowest yields of wheat and spring barley (Tables 4 and 5). Calculated as a weighted average the grain yields on organic dairy farms and organic arable farms were 81 and 58%, respectively, of the conventional level (data not shown).

The amount of N in manure and artificial fertilizer imported to the farm types varied inversely with the livestock units (LU ha^{-1}). For example 107 kg artificial fertilizer N ha^{-1} and 6 kg manure–N was imported to farm type 4 (dairy farm type on sandy loam soil, 0.9 LU ha^{-1}), and only 89 kg artificial fertilizer N ha^{-1} and no manure–N was imported to farm type 5 (dairy farm type on sandy loam soil, 1.7 LU ha^{-1}). These differences are due to the strict Danish regulations on the use of manure and limitations in the use of fertilizer (Hutchings et al., 2005).

Generally the pig farm types imported less artificial fertilizer P (2–9 kg P ha^{-1}) compared with other farm types (10–114 kg P ha^{-1}), because pig manure has a high P content. Pig farm types used more heating oil than any of the other farm types due to the heating requirements for animal houses. There was a tendency for higher diesel imports to dairy farm types in comparison to other farm types. This was caused by the high diesel requirements for the processing and handling of roughage for feeding. The expenditure on chemicals (per ha) was highest for the two farm types producing sugar beets and potatoes. This was in agreement with our expectations, since cultivation of sugar beets and potatoes often includes high levels of pesticide use (Christensen and Huusom, 2003). The data in the farm accounts on chemicals purchased were not specified, but it was assumed that most of these chemicals were pesticides, although some may have been detergents for cleaning pig housing and milking equipment. The use of chemicals was lowest on the dairy farm types, probably because of low pesticide use for grassland.

Product sales from the farm types are not shown in Tables 4 and 5, but are presented at the open database (Nielsen et al., 2003).

3.3. Nutrient balances and emissions from selected farm types

The selected farm types together do not represent the entire Danish agricultural sector and therefore the results are solely valid for the farm types presented in Tables 4 and 5.

The N and P farm gate balances (surpluses) and the emissions from selected farm types on sandy loam soil and sandy soil are presented in Tables 6 and 7, respectively. Tables 6 and 7 shows that dairy farm types have the highest N and P surplus per ha followed by pig farm types and cash

Table 6
Emissions per year from selected farm types on sandy loam soil

Farm type Main product	2 Sugar beets	3 Grass seeds	4 Milk	5 Milk	7 Organic milk	8 Pig	9 Pig	11 Cash crops
Livestock density (LU ha^{-1})	0.5	0.2	0.9	1.7	1.2	0.7	1.6	0.1
N surplus (kg N ha^{-1})	70 ± 13	75 ± 13	137 ± 11	204 ± 16	80 ± 17	114 ± 15	142 ± 18	80 ± 14
P surplus (kg P ha^{-1})	3	4	7	15	1	12	21	5
Emissions								
Nitrate (kg N ha^{-1})	23	34	68	90	6	70	63	48
Ammonia (kg N ha^{-1})	20	15	27	44	23	27	43	11
Nitrous oxide (kg N ha^{-1})	3.2	3.2	5.7	8.3	4.5	4.5	4.5	3.8
Phosphate (kg P ha^{-1})	0.3	0.4	0.7	1.5	1.2	1.2	2.1	0.5
Methane (kg CH_4 ha^{-1})	21	8	101	181	17	17	34	3
CO_2 fossil (t CO_2 ha^{-1})	79	66	76	86	90	90	113	97

±, the standard deviation.

crops farm types. Using the estimated standard errors organic dairy farm types have lower N surplus than conventional, and conventional dairy farm types with high livestock density had higher N surplus than the other livestock farm types. There were probably no differences between different cash crop farm types.

For more details on the N nutrient balances see Kristensen et al. (2005a). Following our methodology, Tables 6 and 7 show the same differences in the emissions between farm types, with comparably high ammonia emissions on high livestock density pig and dairy farm types and low emissions on organic farm types.

The emission per ha of nutrients (nitrate, ammonia, nitrous oxide and phosphate) was higher on the farm types with the highest livestock density, as also shown in Kristensen et al. (2005a). The cash crop farm types (e.g. farm types 2, 3 and 11) in general emitted less nutrients than livestock farms. This is because manure–N is utilised less efficiently compared with N in artificial fertilizer and because the storage and handling of manure result in losses of ammonia and nitrous oxide. The emission of phosphate was also higher from farm types with a high livestock density.

The emissions of nitrate and phosphate were generally lower from farm types on sandy loam soil than farm types on sandy soil. This could be explained by the higher crop yields (shown in Tables 4 and 5) due to higher soil fertility and more stable water supply on sandy loam soil. Moreover, the precipitation surplus is generally higher in the western part of Denmark where most sandy soils are situated resulting in a higher risk of leaching during winter.

Conventional dairy farm types in general had higher nitrate emissions (68–108 kg N ha^{-1}) compared with pig farm types (63–95 kg N ha^{-1}). The higher nitrate emissions were caused by a lower N efficiency in the milk/meat production compared with pork production and the fact that fewer cash crops are sold from dairy farms than pig farms. The phosphate emissions from dairy farm types were lower (0.4–1.7 kg P ha^{-1}) compared with pig farm types (1.2–2.2 kg P ha^{-1}). In Danish legislation the use of N is limited but not the use of P and this probably causes heavier P fertilization on pig farms than on dairy farms, simply because more P is applied per unit N. Pig manure has a higher P/N ratio than cattle manure (Poulsen and Kristensen, 1998). There were no differences in the ammonia emissions per ha between dairy farm types and pig farm types at comparable livestock density. The methane emission per ha was higher from dairy farm types (101–189 kg CH_4 ha^{-1}) compared with all other farm types (3–63 kg CH_4 ha^{-1}), due to the enteric fermentation of cattle.

In general the conventional dairy farms emitted more nitrate (68–108 kg NO_3–N ha^{-1}), ammonia (27–44 kg

Table 7
Emissions per year from selected farm types on sandy soil

Farm type Main product	15 Potatoes	16 Milk	17 Milk	19 Organic milk	20 Pig	21 Pig	23 Beef	24 Grain
Livestock density (LU ha^{-1})	0.4	1.1	1.8	1.4	0.8	1.5	0.7	0.0
N surplus (kg N ha^{-1})	103 ± 19	150 ± 12	209 ± 17	110 ± 23	107 ± 14	148 ± 19	136 ± 11	78 ± 14
P surplus (kg P ha^{-1})	8	9	18	4	14	23	5	7
Emissions								
Nitrate (kg N ha^{-1})	84	85	108	32	77	95	86	73
Ammonia (kg N ha^{-1})	18	30	43	27	28	44	23	10
Nitrous oxide (kg N ha^{-1})	4.5	7.0	8.9	5.7	4.5	5.1	6.4	3.8
Phosphate (kg P ha^{-1})	0.8	0.9	1.7	0.4	1.4	2.2	0.5	0.7
Methane (kg CH_4 ha^{-1})	26	113	189	141	19	33	63	3
CO_2 fossil (t CO_2 ha^{-1})	70	74	85	64	79	111	58	55

±, the standard deviation.

NH_3–N ha^{-1}), and nitrous oxide (5.7–8.9 kg N_2O–N ha^{-1}), compared with organic dairy farms (6–32 kg NO_3–N ha^{-1}, 23–27 kg NH_3–N ha^{-1} and 4.5–5.7 kg N_2O–N ha^{-1}). The very low nitrate emission (6 kg NO_3–N ha^{-1}) from farm type 7 (organic dairy farm on sandy loam soil) was a result of low N surplus (80 kg N ha^{-1}), high denitrification and high accumulation of N in the soil. Further results regarding the N surplus of different farm types are presented by Kristensen et al. (2005a).

Emission of fossil CO_2 was a function of combustion of both heating oil and diesel. So even though the pig farm types used less diesel per ha than dairy farm types, the smaller amount of CO_2 from the diesel combustion was counterbalanced by a higher emission of CO_2 from the combustion of heating oil.

4. Discussion

4.1. Methodology: establishment of farm types

The results presented demonstrate how the resource use and emissions from farm types can be modelled on the basis of representative farm accounts and accepted norms for feed conversion and fertilization.

An important strength of the method is that the farm types are representative, partly because of the use of the representative data set of farm accounts and partly because of the adjustment to the national level statistics. The farm types are based on realistic and documented levels of resource use per unit agricultural product and the emissions, therefore, reflect average production levels and efficiency within different farm types. The farm types are all consistent in terms of crop–livestock interactions, and together they form the national agricultural model that documents the total resource use and emissions of the Danish agricultural sector, including the exchange of manure and straw between farm types.

The method of establishing a set of representative farm types including emissions may be used in many European countries, because similar data sets are reported yearly to Eurostats' FADN. However, the exact modelling approach must be adjusted according to local regulations and N and P norms and the level of detail recorded. For some countries, e.g. Netherlands, data on inputs required for agricultural production are given in quantities instead of expenditures (Poppe and Meeusen, 2000) and thereby the establishment of farm types is facilitated.

The major drawback of the method from our point of view is that the large variation that may exist between farms within one farm type in e.g. feed or fertilizer use efficiency due to differences in farm management skills and strategic choices of crop rotation and feed planning is not reflected in differences between the farm types. This was, however, a necessary choice based on the primary purpose of establishing representative and coherent farm types that could be used for environmental assessment of farm types and agricultural products.

According to Poppe and Meeusen (2000) and EEA (2005), there are two important shortcomings of the FADN, namely, that fertilizer purchases are only valued in monetary units and that it is not disaggregated into the different units such as N, P and K. In the development of the farm types a lot of effort was devoted to estimating the N flows and we did it on the basis of the Danish legislation on fertilizer use. Even so we underestimated the N fertilizer use by 10%. This could have been avoided if we had had the figures for the different volumes of N, P and K fertilizers purchased for the different farm types. So we can only support the request from the EEA (2005) for the expansion of FADN to include the volume of fertilizers.

The amounts of feed and fertilizer purchased could have been modelled for individual farms based on the monetary information using standard prices per unit. But that might have introduced another bias because of differences in the actual price paid per unit, for example farms that get discount prices, would in reality have used more feed or fertilizer than estimated from average prices. Furthermore Halberg et al. (2000) found that calculation of purchased fertilizer from prices is very sensible to unit prices.

Another drawback is the relatively large number of small co-enterprises in the Danish farm types resulting from combining many different co-enterprises (for example two dairy farms growing 5 ha with cash crops, one bread wheat, the other sugar beets will result in a combined type growing 2.5 ha of each). This issue may not be a problem in regions with more specialized farm types compared with the Danish mixed dairy farms.

The national agricultural model did not initially account for the total use of artificial fertilizer N, diesel and heating oil in Danish agriculture, why correction factors were used. While this secures consistency with national level statistics, it is not a completely satisfactory solution because the error may in fact be due to an underestimation in specific rather than all farm types.

The statistical weighting method used to divide the farm accounts into farm types ensures representativity of each farm type. However, due to this method it was not possible to test statistically the variation between farm types. Therefore, the uncertainty of the N balances was estimated using the variance of the individual inputs (e.g. fertilizer, feed) and outputs (e.g. cash crops, milk) (see methods).

Using this method it was established that dairy farm types had higher N surpluses and losses compared with pig and cash crop farm types. There was higher N surplus on conventional farm types compared to organic farm types, and pig and dairy farm types with high livestock density had higher N losses compared to farm types with lower livestock density.

Using the coefficients of variation established for individual farms on the model types, which are based on averages of a large number of farms, probably overestimates the variation. Therefore, we find that these are conservative estimates and it seems therefore reasonable to use these estimates to assess differences in the emissions also. The findings were in agreement with studies on Danish pilot farms (Halberg et al., 1995; Halberg, 1999; Nielsen and Kristensen, 2005).

In a parallel study by Knudsen et al. (in press) sensitivity analyses were performed to test how the N farm gate balances of the dairy farms were affected by changes in amount of BNF, N efficiency in dairy herd, crop yields and N in fodder produced on the farm. The sensitivity analyses showed that the N farm gate balances on conventional farms had a relatively low degree of sensitivity to changes in assumptions (Knudsen et al., in press), due to small areas grown with fixating crops. On organic farms the sensitivity was higher. The N farm gate balances of the organic dairy farms increased 17–19% if a 25% higher BNF was assumed. The N farm gate balances of conventional dairy farms decreased by 5–8% only with 10% increased N content in home grown fodder. On this basis we conclude that the overall pattern and level of nutrient balances and related emissions on the farm types have a satisfactory degree of precision. For further details see Knudsen et al. (in press).

In this study the national agricultural model was used to demonstrate resource uses and emissions of different farm types, but it can also be used on a sector level. For example the eight dairy farm types and six pig farm types can be aggregated (by using population values equal to those in Tables 1 and 2) into two farm types representing the specialized dairy sector and specialized pig sector, respectively. The specialized dairy sector then accounts for 86% of the milk produced in Denmark and the specialized pig sector then accounts for 76% of the fattening pigs produced in Denmark. The establishment of these specialized sectors can give information on which sector is main contributor of different emissions, and which sector has the highest resource use.

4.2. Farm nutrient balances and emissions

Nielsen and Kristensen (2005) studied data from 56 Danish livestock farms collected from 1997 to 2003 and found that N and P surplus increased significantly with increasing livestock density. This is in good agreement with our results showing the emissions of nutrifying substances in general were higher from farms with high livestock density. Holbeck and Hvid (2004), Hvid (2005a,b) and Kristensen et al. (2004) analysed data from farms from 1999 to 2003, and found that the N surplus was higher from dairy farms in comparison with pig farms. This is also in accordance with our results and the results of Nielsen and Kristensen (2005).

In another study based on the national agricultural model the FADN data from 1999 was updated by FADN data from 2002 (Kristensen et al., 2005b). The study showed that the nitrate leaching per hectare was 63 kg N. Exactly the same result was obtained in the 'Evaluation of the Action Plan for the Aquatic Environment II' (Action Plan for the Aquatic Environment, 2003), where the nitrate leaching per hectare for the year 2002 was estimated. Thus nitrate leaching estimated by the national agricultural model is in good agreement with the 'Evaluation of the Action Plan for the Aquatic Environment II'.

Watson et al. (2002) estimated the N and P surpluses on eight Swedish dairy farms (livestock density higher than 0.8 LU ha^{-1}) to 42–128 kg N ha^{-1} and 1–13 kg P ha^{-1}, respectively. Our results on N and P surpluses from organic dairy farms were within these intervals. Verbruggen et al. (2005) estimated the N surplus on conventional specialized dairy farms in 2001 in Flanders to 238 kg N ha^{-1}, which is high compared to our dairy farm types (137–209 kg N ha^{-1}). But here it must be taken into considerations that the livestock density in the study from Flanders was 2.98 LU ha^{-1}, which also is high compared to our dairy farm types. In a study by Haas et al. (2001) the average N and P surplus on German conventional dairy farms (2.0 LU ha^{-1}) were 80 kg N ha^{-1} and 5 kg P ha^{-1}, respectively. These values are low compared to our study, primarily because of lower import of fertilizer and feed to the German farms, which have very high yields on non-fertilized grass leys.

4.3. Environmental indicators

The FADN contains a lot of information, which until now has mainly been used for economic purposes. However, as our study demonstrates the FADN data could also be used to develop more agri-environmental indicators, which give insight into the environmental impact caused by the agricultural sectors in European countries.

The European Environmental Agency (EEA, 2005) recently developed and evaluated agri-environmental indicators for monitoring the integration of environmental concerns into the Common Agricultural Policy. EEA (2005) characterises FADN as the only harmonised micro-economic database that combines data on farm structure, input use and economic variables and FADN is described as a valuable data source for the establishment of agri-environmental indicators to describe energy use, cropping/livestock patterns and organic farm incomes. The FADN was for example used to develop a farm typology to explain general trends in intensification/extensification. However, EEA (2005) does not mention FADN as a potential data source for establishment of agri-environmental indicators describing the emissions of methane, nitrous oxide and ammonia.

Much of the information (e.g. agricultural area, crop yields, livestock density) from the farm accounts which we used for the national agricultural model is also included in the FADN, and therefore it might be possible to use the FADN directly to get an insight into the environmental impact of the agricultural production. Brouwer et al. (1995)

used data from the FADN to assess the N surpluses at farm level in the European Union and found that N surpluses varied widely across groups of farms in the EU because of the differences in farm structure and input use. Fais et al. (2005) developed a methodology where FADN data were combined with statistical, administrative and cartographic information, and then by the use of geographic information systems (GIS) technology it was possible to produce and organise data at geographical level within a region in Italy. Thereby data for environmental indicators (e.g. fertilizer consumption, soil erosion risk) could be spatial referenced and used for agri-environmental analysis in specific regions.

Other types of area-based environmental indicators can also be obtained from the national agricultural model; for example the use of non-renewable resources as heating oil, diesel and artificial fertilizer P (Tables 4 and 5). The purchase of chemicals (monetary units per ha) can also be used as an area-based environmental indicator, but a major disadvantage is that the chemicals are not specified and therefore their toxicity and chemical/physical properties are unknown. The percentage of agricultural area with permanent grass and set-aside indicates the amount of extensively used agricultural area, and can therefore also be used as an environmental indicator. On the other hand data from FADN cannot be directly used for estimating agri-environmental indicators such as soil quality and biodiversity.

The environmental indicators presented in this paper are all area-based. There is an increasing interest in product-based environmental assessments (LCA) because there is a need to evaluate global emissions and impacts from the whole production chain in relation to types and amounts of products consumed (Haas et al., 2000; de Boer, 2003; Halberg et al., 2005). Product-based environmental assessments of agricultural products based on the data from the national agricultural statistics have been published by Nielsen et al. (2003) and Dalgaard and Halberg (2005).

5. Conclusion

On the basis of a representative sample of farm accounts collected and processed for agricultural statistics and for reporting to FADN, a national agricultural model has been established that can provide data of resource use, production and environmentally important emissions for a set of representative farm types. Within each farm type there was a consistent relation between resource use, production and emissions, and all 31 farm types cover the entire Danish agricultural sector.

Production data and resource use (e.g. soybean meal, diesel, artificial fertilizer N) for all farm types were aggregated and by comparison they were shown to be in good agreement with Danish National Statistics, except for artificial fertilizer N, diesel and heating oil where it was necessary to use correction factors to reach the same level as the Danish National Statistics. Thereafter the national agricultural model could be used for delivering data from the 31 coherent and representative farm types for area-based and product-based (LCA) environmental assessments.

Results (per ha) showed that pig farm types imported more heating oil and less artificial fertilizer P compared to other farms, and dairy farms had the highest consumption of diesel. N in manure and artificial fertilizer imported to the selected farm types varied inversely with the livestock density.

Results (per ha) on emissions from the selected farm types showed that the emissions of nutrients (nitrate, ammonia, nitrous oxide and phosphate) in general were higher on the farm types with the highest livestock density. Conventional dairy farm types in general had higher nitrate emissions but lower phosphate emissions compared with pig farm types. The methane emission was higher from dairy farm types compared with all other farm types. In general the conventional dairy farms emitted more nitrate, ammonia, and nitrous oxide, compared with organic dairy farms.

It can be concluded that the resulting national agricultural model successfully establishes a method of modelling coherent and representative farm types based on generally available data. This method will then facilitate the establishment of representative agro-ecological models of typical farms, which can be used for environmental assessments.

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2

LCA of Soybean Meal

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LCA Case Study

LCA of Soybean Meal

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Abstract

Background, Aim and Scope. Soybean meal is an important protein input to the European livestock production, with Argentina being an important supplier. The area cultivated with soybeans is still increasing globally, and so are the number of LCAs where the production of soybean meal forms part of the product chain. In recent years there has been increasing focus on how soybean production affects the environment. The purpose of the study was to estimate the environmental consequences of soybean meal consumption using a consequential LCA approach. The functional unit is 'one kg of soybean meal produced in Argentina and delivered to Rotterdam Harbor'.

Materials and Methods. Soybean meal has the co-product soybean oil. In this study, the consequential LCA method was applied, and co-product allocation was thereby avoided through system expansion. In this context, system expansion implies that the inputs and outputs are entirely ascribed to soybean meal, and the product system is subsequently expanded to include the avoided production of palm oil. Presently, the marginal vegetable oil on the world market is palm oil but, to be prepared for fluctuations in market demands, an alternative product system with rapeseed oil as the marginal vegetable oil has been established. EDIP97 (updated version 2.3) was used for LCIA and the following impact categories were included: Global warming, eutrophication, acidification, ozone depletion and photochemical smog.

Results. Two soybean loops were established to demonstrate how an increased demand for soybean meal affects the palm oil and rapeseed oil production, respectively. The characterized results from LCA on soybean meal (with palm oil as marginal oil) were 721 g CO_2 eq. for global warming potential, 0.3 mg CFC11 eq. for ozone depletion potential, 3.1 g SO_2 eq. for acidification potential, –2 g NO_3 eq. for eutrophication potential and 0.4 g ethene eq. for photochemical smog potential per kg soybean meal. The average area per kg soybean meal consumed was 3.6 m^2year. Attributional results, calculated by economic and mass allocation, are also presented. Normalised results show that the most dominating impact categories were: global warming, eutrophication and acidification. The 'hot spot' in relation to global warming, was 'soybean cultivation', dominated by N_2O emissions from degradation of crop residues (e.g., straw) and during biological nitrogen fixation. In relation to eutrophication and acidification, the transport of soybeans by truck is important, and sensitivity analyses showed that the acidification potential is very sensitive to the increased transport distance by truck.

Discussion. The potential environmental impacts (except photochemical smog) were lower when using rapeseed oil as the marginal vegetable oil, because the avoided production of rapeseed contributes more negatively compared with the avoided production of palm oil. Identification of the marginal vegetable oil (palm oil or rapeseed oil) turned out to be important for the result, and this shows how crucial it is in consequential LCA to identify the right marginal product system (e.g., marginal vegetable oil).

Conclusions. Consequential LCAs were successfully performed on soybean meal and LCA data on soybean meal are now available for consequential (or attributional) LCAs on livestock products. The study clearly shows that consequential LCAs are quite easy to handle, even though it has been necessary to include production of palm oil, rapeseed and spring barley, as these production systems are affected by the soybean oil co-product.

Recommendations and Perspectives. We would appreciate it if the International Journal of Life Cycle Assessment had articles on the developments on, for example, marginal protein, marginal vegetable oil, marginal electricity (related to relevant markets), marginal heat, marginal cereals and, likewise, on metals and other basic commodities. This will not only facilitate the work with consequential LCAs, but will also increase the quality of LCAs.

Keywords: Agriculture; consequential LCA; soybean meal; system expansion

Introduction

Soybean meal is an important input to livestock and fish production globally and comes from the cake of soybeans after crushing the beans and extracting the soybean oil. In 2004, the consumption of soybean meal in the EU25 was 34 million tonnes (Oil World 2005). The amount of soybeans and cake of soybeans traded globally increased from 48 million tonnes in 1985 to 106 million tonnes in 2004. Soybeans exported from USA were the fourth most important agricultural commodity in dollar value traded globally in 2004, with soybeans from Brazil ranking as no. 7 and cake of soybeans from Argentina as no. 10 (FAOSTAT 2006a).

The 6,979 million US dollar worth of soybeans *imported* to China in 2004 topped the FAO's list of agricultural commodities traded globally and four European countries (Spain, Italy, the Netherlands and Denmark) with large pig productions imported 46% of the soybean meal from Argentina, Spain, Italy, the Netherlands and Denmark. The global area cultivated with soybeans has expanded from 38 million hectares in 1975 to 91 million hectares in 2005 (FAOSTAT 2006b), with the major land increases taking place in Argentina and Brazil. In Argentina, the area with soybean increased from 6 million hectares in 1996 to 14.2 million in 2004, of which the transgenic varieties accounted for more than 90%, under no-tillage systems (Pengue 2006), where ploughing is not used. The environmental impacts from soybean production have been addressed in several reports, e.g., Dros (2004), Pengue (2005), Benbrook (2005) and Casson (2003). The combination of no-tillage systems and transgenic Roundup Ready (RR) soybeans has made large-scale soybean cultivation a powerful competitor to other types of land use and has caused both a concentration of land tenure, the conversion of traditional farming systems with pastures and hay fields, cereals and other crops, and deforestation. More than 40% of the increased soybean area in Argentina has come from virgin lands, including forests and savannahs, thus causing losses in biodiversity (Pengue 2006). Likewise, in Brazil, the possibility to obtain cheap credit for a fast-return export crop production has allowed soybean producers to expand in a complex interaction with the increasing cattle production leading to deforestation (Dros 2004). The use of glyphosate in Argentina has increased to more than 45 million kg in 2004, up from 20 million in 2000 and less than 1 million in the beginning of the 90s (Pengue 2006). While glyphosate as an active ingredient was previously considered harmless to humans and warm-blooded animals (Anonymous 1996), new research indicates that some of the formulations used with this type of pesticide may cause health problems for farm workers and negative environmental effects on biodiversity and aquatic life, as discussed by Ho & Cummings (2005) and Ho & Ching (2003).

Soybean production is often part of the system when performing Life Cycle Assessments of different agricultural products. Analyzing the environmental consequences of changes in food consumption (Gerbens-Leenes & Nonhebel 2002) or livestock production systems often involves changes in the demand for soybean meal (Cederberg & Mattson 2000, Eriksson et al. 2004, de Boer 2003, Cederberg & Flysjö 2004, Basset-Mens & van der Werf 2005, van der Werf et al. 2005, Dalgaard & Halberg 2005). This is particularly the case when the LCA is based on the consequential approach where the analysts are looking for the marginal product being used or saved when expanding the delimitations of the investigated system in order to avoid allocation (Nielsen et al. 2004).

Mass or economic allocation has been used to distribute the environmental burden between the soybean meal and the soybean oil in most of the LCAs on livestock products where soybean meal is included. In this article, however, we are seeking to avoid an allocation. Instead, we aim at following the principles of system expansion according to ISO 14044 (2006). This means that the system boundaries of soybean production must be enlarged in order to include the production system of the vegetable oil substituted by the soybean oil.

1 Goal and Scope Definition

More specifically, the objectives are

- to establish a reliable representation of soybean meal production for use in LCAs of European livestock production chains
- to identify the environmental hot spots in the product chain of soybean meal

In order to ensure the usability of the LCA presented for researchers preferring to use allocation to handle co-products, sufficient numbers and figures will be displayed to allow for this.

The purpose of this study is to estimate the environmental consequences of soybean meal consumption, and to provide data on the LCA of soybean meal. This is partly because there is a lack of these data for LCAs on livestock products, and because soy protein could potentially have a significant environmental impact as a consequence of the increasing production of meat products worldwide.

Soybean meal is co-produced with soy oil, and a demand for soybean meal obviously necessitates a production of soybean oil. The production of oil might affect other agricultural product systems, but to what extent, and how can it be quantified? In this article; these issues will be analyzed further.

The functional unit is 'one kg of soybean meal produced in Argentina and delivered to Rotterdam Harbor in the Netherlands'. The Netherlands is the country within Europe that imports the largest amount of oil meals (Oil World 2005).

The impact categories considered include: global warming, ozone depletion, acidification, eutrophication and photochemical smog. Impact categories concerning toxic aspects are not included due to methodological limitations. Land use, impacts on biodiversity and other impacts are not integrated in the present LCA, due to methodological limitations (as discussed by Milà i Canals et al. (2007)), although results on land use (unit: m^2year) are presented.

2 Methods

2.1 LCA approach

Identification and delimitation of the analyzed product system is increasingly seen as being important for the outcome and quality of the LCA (Weidema 2003, Ekvall & Weidema 2004, Schmidt 2004). Two fundamentally different approaches can be used in this respect: the new (consequential) approach and the traditional (attributional) approach. Most existing LCAs are based on the attributional approach, but the tendency is for studies to increasingly use the new (consequential) approach (Thrane 2006, Schmidt & Weidema 2007, Ekvall & Andræ 2006, Cederberg & Stadig 2003, Kim & Dale 2002, Dalgaard & Halberg 2005, Weidema 1999). In the present study, it has been chosen to apply the consequential approach, which has two main characteristics:

- It seeks to model the technology (or processes) actually affected by a change in demand (the marginal technology).
- Co-product allocation is 'systematically' avoided through system expansion.

These characteristics are opposite to attributional LCA, where average technologies (not marginal) are used, and where co-product allocation is often handled by mass or value allocation (Weidema 2003).

In consequential LCA we basically use a 'market oriented' approach to identify the affected technology (or process), also called marginal. We continuously ask: what is affected by a change in demand? For example, when impacts related to electricity input for a certain unit process are considered – the question is: what are the environmental consequences related to a change (typically small) in the demand of electricity in this market? Among the Nordic countries, this is mainly coal or gas-based technologies according to Weidema (2003). Hence, in this case the marginal technology is gas or coal (or a mix). In traditional (attributional) LCA, electricity consumption is often modeled as an average of all electricity sources within the region, but this would then include electricity from, for example, windmills, which produce as a function of the wind speed – not the demand. The same applies to other renewable energy sources such as hydropower, which should be left out of the product system, according to the consequential LCA. Thus, in consequential LCA, only affected technologies (or processes) should be included, and socio-economical considerations should be applied to identify these (Weidema 2003).

In this article, the term 'marginal technology (or process)' refers to the technology or process, which is actually affected by a change in demand. The changes that are considered in this article are small, which means that they do not affect the determining parameters for the overall market situation, i.e., the direction of the trend in market volume and the constraints on and production costs of the products and technologies involved (Weidema et al. 2004).

Concerning the handling of co-product allocation, attributional LCAs have often based allocation on the relative value of the products and co-products, be it mass or other parameters. In consequential LCA, however, this is entirely avoided through system expansion (if technical subdivision of the processes is impossible). System expansion means the inputs and outputs are entirely ascribed to the product of interest (often the main product). Subsequently, the product system is expanded to include the products avoided, i.e., products that are avoided due to the co-products. Accordingly, when performing consequential LCA on soybean meal, the inputs and outputs relate entirely to the soybean meal, but the avoided production of vegetable oil, caused by the co-product soybean oil, is included in the calculations. Because vegetable oil (e.g., palm oil, rapeseed oil) is nearly always co-produced with protein (e.g., palm kernel meal, rapeseed meal), this will introduce another need for system expansion, which again could include a co-production of protein, etc. This never-ending story is described by Weidema (1999) as the so-called soybean-rapeseed-loop. While Weidema at the time assumed rapeseed to be the marginal oil replaced by soybean oil, in this article we will demonstrate and compare the use of this loop principle for LCAs of soybean meal using both palm oil and rapeseed oil as the marginal products to be replaced. Palm oil is chosen because Schmidt & Weidema (2007) presently identified this as the marginal oil. For further details regarding the consequential LCA methodology, see Ekvall & Weidema (2004) and Weidema (2003).

2.2 Method applied for LCIA

Among the different methods available for Life Cycle Impact Assessment (LCIA), we have used the EDIP97 (Wenzel et al. 1997, updated version 2.3). The method has been implemented in the PC-tool SimaPro 6.0 (Pré 2004). The EDIP-methodology has recently been launched in a revised EDIP2003 version (Hauschild & Potting 2005) but, as this new revised version has not yet been implemented in any PC-tool, it was decided to stick to the well-documented and familiar EDIP97 methodology.

EDIP97 also includes human toxicity, eco-toxicity, waste and resource use, but we have chosen not to include these impact categories due to methodological limitations regarding pesticide emissions from agriculture.

2.3 System delimitation

The soybean plant (*Glycine max.)* is a legume, which grows to a height of 120–180 cm (Tengnäs & Nilsson 2003). Soybeans contain approximately 35% protein and 18% oil (Møller et al. 2003, an update from Møller et al. 2000) and are the highest-yielding source of vegetable protein globally (Dros 2004, p. 7). The protein is primarily used for livestock feed after crushing and extraction of the oil, which is mainly used for consumption.

Argentina has become the largest global exporter of soybean cake and is projected to have the highest increase in export until 2014 (FAPRI 2006). Therefore, in this study, soybean meal produced in Argentina is used as the marginal soybean meal. An increase in the demand for soybean meal implies an increase in the production of soybean oil, which would then compete with other vegetable oils on the world market. Following the methodology of consequential LCA, this 'avoided production' of vegetable oil must be included in the LCA of soybean meal (protein).

As mentioned earlier, rapeseed oil was until recently regarded as the marginal oil that was affected when the demand for general vegetable oils changed (Weidema 2003). Recent studies, however, show that palm oil has increased its competitiveness compared to other major oils on the market - rape, soy and sunflower oils (Schmidt & Weidema 2007).

The fatty acid composition of rapeseed, soy, sun and palm oil is not the same. Thus, they are not completely substitutable. However, according to Schmidt & Weidema (2007), the oils are substitutable within the most important applications: frying oil/fat, margarine, shortening and possibly salad oils. One of the co-products from palm oil milling is palm kernels, which are processed into palm kernel oil and palm kernel meal. The applications of palm kernel oil, which is a lauric oil, differ from the most important applications mentioned above. The only other lauric oil on the market is coconut oil. The use of this oil is constrained due to a 5–7 year maturing period before harvesting can begin making it less responsive to fluctuations in market demand (Schmidt &

Weidema 2007). Therefore, changes in the production of palm kernel oil are not considered likely to affect the production of coconut oil. Schmidt & Weidema (2007) argue that palm oil and the co-product palm kernel oil jointly can be considered as the marginal oil on the global market. In the following, 'palm oil' designates a mix of oil from mesocarp of fresh fruit bunches and oil from palm kernels.

As market situations often change from one year to another, we have decided to make two LCAs. One with palm oil as the marginal oil (here called: Soybean meal (PO)), and one with rapeseed oil as the marginal oil (here called: Soybean meal (RSO)).

Avoided production of palm oil and rapeseed oil implies avoided production of palm kernel meal or rapeseed meal, respectively. This avoided production of meal will be compensated for by the production of marginal meal. Thus, a demand for soybean meal does not only result in production of the demanded amount, but also in the production of an extra amount of soybean meal to compensate for the 'missing meal' (palm kernel meal or rapeseed meal) that are missing because of the avoided oil production. The extra amount of soybean meal produced will again cause an avoided production of meal, and this loop will continue. The mass of extra soybean meal produced is very dependent on the protein and energy contents of the ingredients involved, and is therefore not the same in the LCA of soybean meal (PO) as in the LCA of soybean meal (RSO). To demonstrate this difference and to facilitate the LCAs of soybean meal, two loops (based on the concept developed by Weidema (1999)) will be established for the two LCAs of soybean meal: A soybean/palm loop for the soybean meal (PO), and a soybean/rapeseed loop for the soybean meal (RSO). Data on dry matter, oil, protein and energy contents of relevant items in the loops are based on data from **Table 1**. The yields of soybean meal and soybean oil from soybeans and the yields of rapeseed cake and rapeseed oil from rapeseeds are calculated on the basis of these data, taking into consideration that some of the oil from soybeans and rapeseeds form part of soybean meal and rapeseed cake, respectively. The yields of oil and kernels from fresh fruit bunches, and the yields of palm kernel oil and meal from the kernels are based on Malaysian data for 2004 given in MPOB (2005). In the calculations, soybean oil substitutes marginal oil at the ratio of 1 to 1 (by weight). The amount of marginal meal substituted by palm kernel meal or rapeseed cake is estimated according to the protein and energy contents (energy in feed is calculated in Scandinavian Feed Units (SFU), where 1 SFU approximately equals the amount of energy in 1 kg barley). For example, one kg of rapeseed cake (= 0.31 kg protein and 1.1 SFU) substitutes 0.95 kg marginal meal, which is a mix of 0.66 kg soybean meal (= 0.28 kg protein and 0.8 SFU) and 0.29 kg spring barley (=0.03 kg protein and 0.3 SFU). Thus, the amount of protein and energy in the rapeseed cake is equal to the total amount of protein and energy in soybean meal and spring barley. In the calculations, spring barley is assumed to be the marginal feed grain, as proposed by Weidema (2003).

3 Inventory

In the following section, we present the data used to establish the crop production and crop processing in the LCAs. As explained, due to the need for systems expansion, the soybean product system includes the cultivation and processing of oil palms, rapeseed and spring barley. For ease of comparison, the crop data are all presented in **Table 2**, while explanations and references are given in separate sections.

3.1 Agricultural production

Soybeans. Yields of 2,630 kg ha^{-1}, which was the average yield in Argentina 2001/2002, were used, cf. Table 2 (SAGPyA, 2006). At this time approx. 25% of the soybean area was cultivated in a system with two crops (typically wheat and soybeans) per year (Begenisic 2003), giving an average land use of 0.88 ha year (=0.75 + (0.25/2)) for the 2,630 kg beans. Fertilizer, diesel and pesticide use was taken from Begenisic (2003), according to whom approximately 70% of the soybeans at that time were cultivated in a no tillage cropping system (diesel consumption: 35 liters ha^{-1}) using transgenic varieties (RR) and 30% were cultivated in a conventional cropping system (diesel consumption: 60 liters ha^{-1}). The nutrient balance approach (Halberg et al. 1995; Kristensen et al. 2005) was used for estimating nitrate and phosphate leaching. No N-fertilizer was given, leaving the beans to depend on biological nitrogen fixation (corresponding to 132 kg N ha^{-1}, estimated from Peoples et al. (1995)), and available soil N for its N supply. N removed from the field was calculated on the basis of yields (see Table 2) and protein content (see Table 1) of the soybeans, and equaled 152 kg N ha^{-1}. Because more N was removed than applied to the soybean fields, it was assumed that nitrate leaching related to soybeans was insignificant (see Table 2). This is in accordance with Austin et al. (2006), who also concluded that less N is applied than removed from the soybean fields in the Pampas in Argentina. Phosphate adsorbs to soil particles and it was assumed that only 2.9% of the P surplus was leached as phosphate (Dalgaard et al. 2006). N_2O emissions were calculated according to IPCC (2000).

Table 1: Characteristics of items in the soybean/palm loop and soybean/rape loop (Møller et al. (2003), an update from Møller et al. (2000))

	Soybeans	Soybean meal	Palm kernel meal	Rapeseed	Rapeseed cake	Spring barley
Dry matter (DM), %	87	87.5	92	92	89	85
Protein, % of DM	40.8	49.1	16.2	21.6	34.8	10.8
Oil, % of DM	20.8	3.2	10.9	48.0	11.2	3.1
Energy, SFU [a] per kg DM	1.44	1.37	0.86	1.86	1.19	1.12

[a] SFU: Scandinavian Feed Units

Table 2: Inventories for cultivation of 1 hectare of soybean, oil palms, rapeseed and spring barley

	Soybean	Oil palms (fresh fruit bunches) [a]	Rapeseed	Spring barley
Location	**Argentina**	**Malaysia**	**Denmark**	**Denmark**
Yields, tons/ha	**2.63**	**18.80**	**2.83**	**4.90**
Resource use				
Fertilizer (N), kg	0	90	167	123
Fertilizer (P), kg	16	12	24	21
Fertilizer (K), kg	0	134	77	62
Diesel, L	42	64	125	114
Lubricant oil, L	4	0	13	11
Electricity (natural gas), kWh	0	7	23	29
Emissions to water				
Nitrate, kg NO_3	0	83	326	202
Phosphate, kg PO_4	0	0.7	0.6	0.7
Emissions to air				
Ammonia, kg	0	0	12.2	10.5
Nitrous oxide, kg	4.7	6.5	6.7	4.8
Nitrogen dioxide, kg	0	1.7	0	0
Sulfur dioxide, kg	0	0.8	0	0

[a] Data based on Yusoff & Hansen (2007)

Average pesticide application over the RR and conventional systems was estimated from Begenisic (2003) and Benbrook (2005). In the no-tillage system, farmers use 5–6 liters of glyphosate solution per ha and an average 0.35 liters of 80% 2,4-D while, in the conventional cropping system, 2 liters of glyphosate is supplemented with 1 liter of imazethapyr. Benbrook (2005) reports an increased used of imazethapyr in Argentina, even though the proportion of RR soybeans has increased, which indicates that this herbicide may be used in combination with glyphosate, possibly to avoid problems with glyphosate-resistant weeds. Imazethapyr is slightly hazardous in WHO (World Health Organization) terminology (Agrocare 2002). The substance has been withdrawn from the European market (Anonymous 2002), but is used in Brazil and Argentina. A number of insecticides are used in soybean cultivation, mostly pyrethroids (0.1 liter per ha of cypermethrin or deltamethrin) and chlorpyrifos (0.8 liter per ha), which are all highly toxic to aquatic environments. The first Sorghum halepensis biotype resistant to glyphosate in the north of Argentina was reported in 2005 (Anonymous 2006a). Because of lack of a reliable fate model to represent the pesticide application techniques used and linking this with the geographical distribution of biodiversity and water bodies, the pesticides were not included in the LCA as such. As discussed below, there is a risk of significant and large-scale impacts on biodiversity because of glyphosate's broad-spectrum effect on non-target plants and amphibians.

Fresh fruit bunches from oil palms. Yields of 18,800 kg ha^{-1} fresh fruit bunches are used, calculated as an average of the yields in 2003 and 2004 as reported by the Malaysian Palm Oil Board (MPOB 2005). Remaining data are based on Yusoff & Hansen (2005). Due to lack of data, MgO fertilizer is not included in the calculations. Production of organic fertilizer is not incorporated in the calculations, because it is assumed that these organic fertilizers are residues that are not produced as a consequence of palm oil cultivation. In accordance with phosphate leaching from soybeans, rapeseed and spring barley, 2.9% of the P surplus is assumed to be leached as phosphate. Literature on nitrous oxide (N_2O) emissions from palm oil cultivation was not available, thus it was assumed that the N_2O emissions were equal to the emissions from soybean cultivation (4.7 kg N_2O ha^{-1}) plus N_2O emissions from the 90 kg N fertilizer that was applied yearly. N_2O emissions from fertilizer were calculated in accordance with IPCC (2000), and the total emission was therefore calculated at 6.5 kg N_2O ha^{-1}. Available information on pesticide use for oil palm cultivation is limited, but according to Wakker (2005), around 25 different pesticides are used and the most commonly used weed killer in oil palm plantations is paraquat dichloride ('paraquat'). Paraquat is banned or restricted in Denmark, Austria, Finland, Sweden, Hungary and Slovenia because of its high toxicity (Anonymous 2006b). Malaysia, the biggest producer of palm oil, has implemented a 2-year phase-out period, but is now reconsidering the phase-out. Glyphosate is also used in oil palm plantations (DTE 2005).

Rapeseed and spring barley. Yields of 2,830 and 4,900 kg ha^{-1}, respectively, were used. All agricultural data are from a National Agricultural Model (Dalgaard et al. 2006), which is representative for the Danish agricultural sector. The model consists of 31 farm types that are representative of the entire agricultural sector in Denmark. For each farm type, resource use and emissions are established using representative farm accountancy data. The Economic model ESMERALDA (Jensen et al. 2001) was used to identify the marginal rapeseed and spring barley producers amongst the 31 farm types, so that marginal Danish data and not average data were used. In the National Agricultural Model, nitrate leaching was assumed to be equal to the farm-gate N balance minus ammonia losses and denitrification (Kristensen et al. 2005) and net change in soil N status. The farm-gate N balance was established according to the methods developed by Halberg et al. (1995) and Kristensen et al. (2005). N_2O emissions were calculated according to IPCC (2000), and the diesel use was modeled according to Dalgaard et al. (2001). The balance approach was also used for calculation of phosphate leaching, but assuming that only 2.9% of the P surplus was leached (Dalgaard et al. 2006). Pesticide use was estimated from inventories established yearly by the Danish Environmental Protection Agency (Anonymous 2005) based on total national sales and the distribution of crops. The most abundant herbicide in rapeseed was clomazone (used on 43% of the rapeseed area in 2004, giving on average 0.14 liter per ha) followed by propyzamide (1 liter per ha on 36% of the land, high aquatic toxicity) and clopyralid (0.2 liter per ha on 17% of land). Approximately 60% of the cropped rapeseed land was treated with insecticides, most often 0.2 liter per ha of cypermethrin, which is classified as moderately dangerous for humans and highly toxic to aquatic organisms. A large variety of herbicides, fungicides and insecticides were used in cereal production and the average number of standard approved dosages used per ha was 1.4 herbicide treatments, 0.61 for fungicides, 0.27 for insecticides and 0.12 for growth regulator applications (Anonymous 2005). The most frequently used herbicide applied to 36% of the barley area was tribenuron-methyl, which is classified as moderately dangerous for humans and highly toxic to aquatic organisms.

LCA data on material inputs. Within Europe a change in demand for artificial fertilizer affects the less competitive fertilizer producers, as the European market has experienced a decrease in the consumption of fertilizer due to environmental restrictions (Weidema 2003, p. 73). Data on artificial fertilizers (nitrogen, phosphorous and potassium) are from Patyk & Reinhardt (1997), as these data are assumed to represent the less competitive technology. Due to lack of data, the same data are assumed to be valid for palm oil production in Malaysia. Data on the use of agricultural machinery are based on Borken et al. (1999), but moderated to average load. These data are average, but are assumed not to differ from marginal data. For further information on the above-mentioned data, go to www.lcafood.dk (Nielsen et al. 2003). Data on electricity (electricity from fuel gas power plant in the Netherlands), pentane (used instead of hexane), transport by truck (28 t), heat (oil) and heat (gas) are all from the Ecoinvent Centre (2004).

3.2 Milling plants

The inventories for processing of soybeans, rapeseeds, fresh fruit bunches from oil palms and palm kernels are presented in **Table 3**. Fossil energy related emissions are not shown. Based on data from Oil World (2005) it is assumed that the losses from all milling processes are 2%.

Table 3: Inventories for millings plants. Functional unit: Processing of 1 ton soybeans, fresh fruit bunches, palm kernels and rapeseeds, respectively

	Soybean mill	Palm oil mill	Palm kernel mill	Rapeseed mill
Location	**Argentina**	**Malaysia**	**Malaysia**	**Denmark**
Products	**Soybean meal Soybean oil**	**Palm oil Palm kernels Pulp**	**Palm kernel oil Palm kernel meal**	**Rapeseed meal Rapeseed oil**
Transport				
Transport to mill (28 t lorry)	500 km	0 km	150 km	150 km
Transport to mill (tractor)		22 MJ Diesel		
Resources				
Hexane	0.40 kg	0	1.99 kg	0
Diesel for machinery			32 MJ	
Electricity (natural gas)	12 kWh		68 kWh	50 kWh
Heat (oil)	145 MJ	–	335 MJ	340 MJ
Heat (gas)	282 MJ	–	–	
Emissions to air				
Methane		9,570 g		
Hexane	0.20 kg	–	1.99 kg	
Carbon monoxide	Energy related	50 g	Energy related	Energy related
Nitrogen oxides	Energy related	120 g	Energy related	Energy related
NMVOC, volatile organic compounds	Energy related	239 g	Energy related	Energy related
Sulfur dioxide	Energy related	435 g	Energy related	Energy related
Particles	Energy related	276 g	Energy related	Energy related
Emissions to water				
BOD5, Biological Oxygen Demand	17 mg			
COD, Chemical Oxygen Demand	61 mg			
Nitrate	4 mg	182 g		

Soybean mill. Soybean meal consumed in the EU is primarily milled outside the EU (Oil World 2005). Consequently, it is assumed in this study that it is milled where it is produced (Argentina). The amount of hexane used for oil extraction and emitted is based on Cederberg (1998), and energy use and emissions to water are from Reusser (1994). It is assumed that soybean meal is sailed 12,082 km from Argentina (Rosario Harbor) to the Netherlands (Rotterdam Harbor). According to Oil World (2005), the Netherlands is the largest importer of soybean meal in Europe.

Palm oil mill. Processing data are based on palm oil mills owned by Unilever in 1990 (Unilever 2004). The fresh fruit bunches are transported by tractor as the oil mills are always placed near the oil palm plantation in order to have a relatively short transport time to avoid decomposition of fatty acids in the fresh fruit bunches. Energy in the palm oil mill is supplied by incineration of empty fruit bunches, mesocarp fibers and nut shells. All airborne emissions given in Table 3 are related to storage and burning of this organic matter. The emissions to air and N to water are based on Zah & Hischier (2003). Palm oil is extracted without the use of organic solvents. Possible empty fruit bunches, mesocarp fibers and shells that are not used for energy production are not included.

Palm kernel oil mill. Similar to palm oil mill, processing data for the palm kernel oil mill are based on Unilever (2004). The oil is extracted using hexane as a solvent.

Rapeseed mill. Materials and energy used for processing of rapeseeds are from Scanola, a Danish mill that processes approximately 220,000 tonnes of rapeseeds annually (Emmersen 2005). In contrast to the soybean and palm kernels mill, organic solvent is not used for the extraction.

4 Results

4.1 Soybean loops

The aim of this section is to demonstrate how an increased demand for soybean meal affects the agricultural production using the two soybean loops (PO and RSO).

Fig. 1 shows the soybean meal loop with palm oil as a marginal oil. The loop is divided into three parts that are interrelated in a loop as illustrated by the large bold arrows. To produce 1,000 g soybean meal, 1,210 g soybean is needed. These soybeans contain sufficient oil to produce 191 g pure soybean oil leaving 29 g oil in the soybean meal (not shown). The soybean oil is sold on the market and assumed to substitute palm oil, which is a mix of palm oil (from mesocarp) and palm kernel oil. When the fresh fruit bunches are milled, there is a large co-production of organic residues that are used to produce energy for the palm oil mill (see also Table 3). For each 1,000 g of soybean meal produced, there is an avoided production of 23 g palm kernel meal, which is substituted by marginal meal. The marginal meal is soybean meal but, as the protein and energy content is higher in soybean meal (see Table 1) than in palm kernel meal, this is compensated for by a mix of soybean meal and spring barley (23 g of palm kernel meal contains the same amount of protein and energy as a mix of 5 g of soybean meal and 12 g spring barley). Consequently, after the first turn in the loop, which was caused by an increased demand for 1,000 g of soybean meal, the production has increased to 1,005 g. By making this iteration for each turn, the extra amount of soybean meal produced is getting smaller. The iteration was carried out in the LCA-tool SimaPro (Pré 2004), and the result showed that an increased demand for 1,000 g of soybean meal caused a production of 1,005 g of soybean meal, –856 g of fresh fruit bunches and 12 g of spring barley.

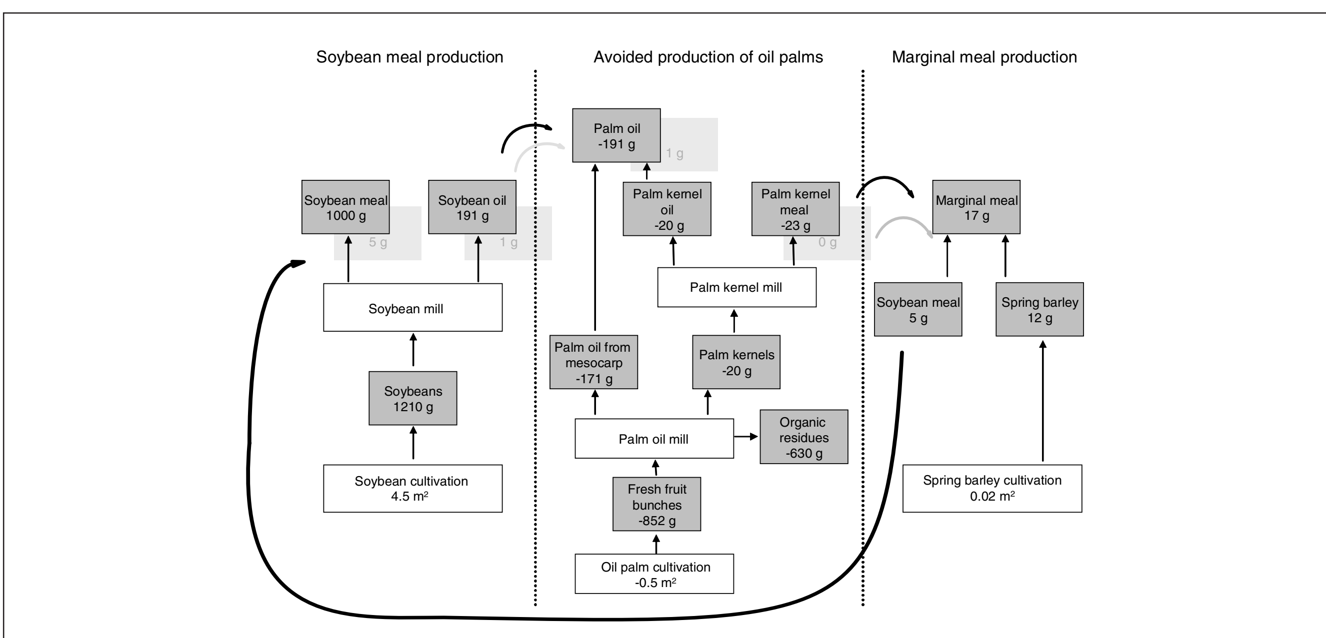

Fig. 1: Soybean/palm loop for LCA of soybean meal (PO). First turn in the loop: An increased demand for 1,000 g of soybean meal results in production of 1,005 (=1,000 + 5) g soybean meal, –852 g fresh fruit bunches and 12 g spring barley. Shadowed boxes show the beginning of second loop

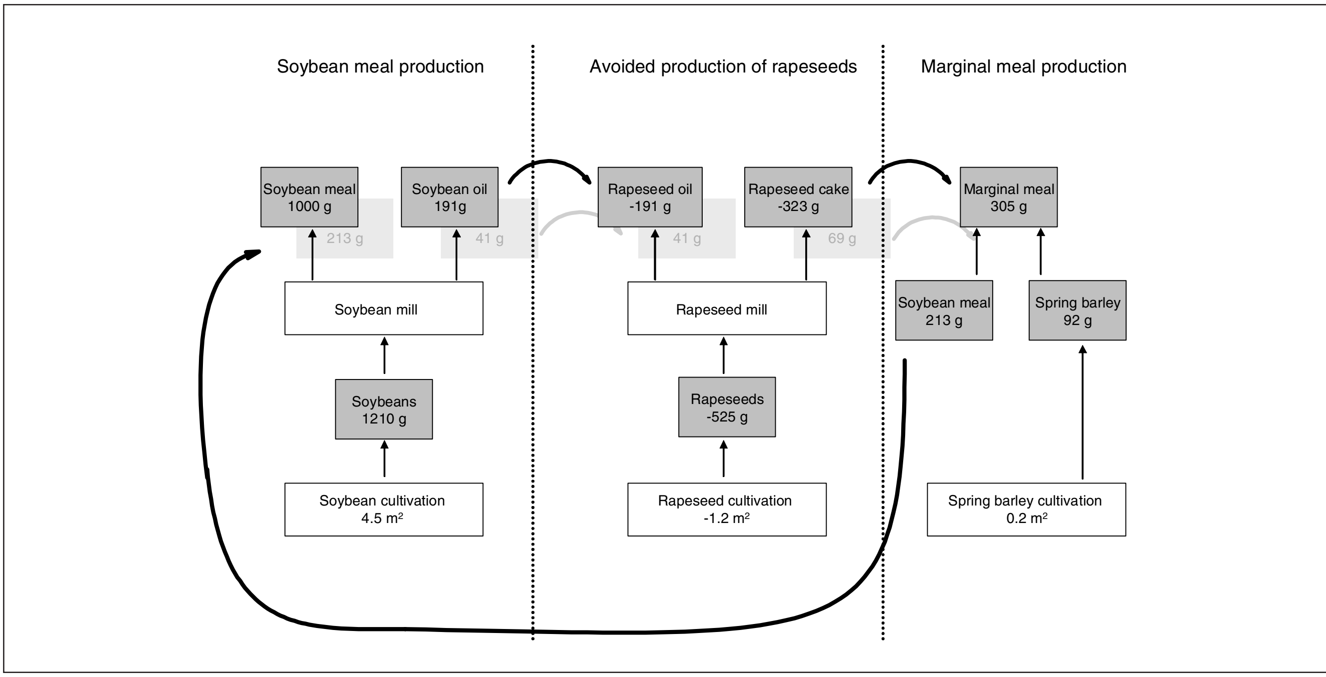

Fig. 2: Soybean/rapeseed loop for LCA of soybean meal (RSO). First turn in the loop: An increased demand for 1,000 g of soybean meal results in production of 1,213 (=1,000 + 213) g soybean meal, –525 g rapeseeds and 92 g spring barley. Shadowed boxes show the beginning of second loop

Fig. 2 shows the soybean/rapeseed loop. Here the soybean oil is assumed to substitute rapeseed oil on the market. For each 1,000 g of soybean meal produced, there is an avoided production of 323 g rapeseed cake, which is considerably more than the 23 g palm kernel meal in the soybean/palm loop (see Fig. 1). Rapeseed cake contains more protein compared with palm kernel meal, thus the soybean meal/spring barley ratio is lower in the soybean/palm loop (see Fig. 1) compared with the soybean/rapeseed loop (see Fig. 2). By iteration, the amount of soybean meal produced can be calculated as for the soybean/palm loop. The increased demand for 1,000 g of soybean meal causes a production of 1,271 g of soybean meal, –667 g rapeseed and 117 g spring barley.

Once the soybean loops are established, the effect of the increased demand for soybean meal on the palm oil, rapeseeds and spring barley needed can be quantified and used in the LCAs of soybean meal (RSO) and soybean meal (PO). Results from the LCAs are presented in the following.

4.2 Characterized results

Table 4 shows the characterized results of the two soybean meal LCAs from 'Rotterdam Harbor' together with the 'from farm gate products' used. Soybean meal (RSO) has a lower environmental impact for all effect categories (except photochemical smog) compared with soybean meal (PO), and this can be ascribed to the fact that the avoided environmental impact from rapeseeds is much larger compared with the avoided environmental impact from palm oil ('fresh fruit bunches from farm gate'). It is worth noting that exactly the same process for soybean cultivation is used for the two soybean meal productions.

In **Table 5**, the economic and mass-allocated, characterized results are presented. A comparison between soybean meal (PO) (see Table 4) and economically allocated soybean meal (69%) (see Table 5) shows that the characterized results for the impact categories 'global warming', 'ozone depletion' and 'acidification' are very similar. 'Eutrophication' from soybean

Table 4: Characterized results of soybean meal (PO), soybean meal (RSO) and crops involved in the life cycle of soybean meal. Functional unit: 1 kg of product

	Unit	Soybean meal (PO)	Soybean meal (RSO)	Soybeans	Fresh fruit bunches	Rapeseeds	Spring barley
Delimitation		**from Rotterdam**		**from farm gate**			
Global warming	g CO_2 eq.	721	344	642	177	1,550	671
Ozone depletion	mg CFC11 eq.	0.27	0.20	0.08	0.02	0.23	0.12
Acidification	g SO_2 eq.	3.1	–1.2	0.8	1.6	11.8	5.8
Eutrophication	g NO_3 eq.	–2	–81	1	8	139	53
Photochemical smog	g ethane eq.	0.4	0.4	0.1	0.0	0.3	0.2

Table 5: Characterized results of soybean meal. Calculated by the use of economic and mass allocation. Functional unit: 1 kg of soybean meal (PO) delivered to Rotterdam Harbor

	Unit	Economic allocation [a]		Mass allocation	
		Soybean meal (69%)	Soybean oil (31%)	Soybean meal (84%)	Soybean oil (16%)
Global warming	g CO_2 eq.	726	1,819	901	901
Ozone depletion	mg CFC11 eq.	0.20	0.49	0.24	0.24
Acidification	g SO_2 eq.	3.3	8.3	4.1	4.1
Eutrophication	g NO_3 eq.	3.1	7.8	3.8	3.8
Photochemical smog	g ethane eq.	0.3	0.8	0.4	0.4

[a] Prices from Argentina year 2002: Cake of soya beans 158 US$, oil of soya beans 396 US$ (FAOSTAT 2006)

meal (69%) (see Table 5) is positive, in contrast to soybean meal (PO) (see Table 4), but still much lower than 'eutrophication' from 'rapeseeds' and 'spring barley' in Table 4.

Normalization of the characterized results in Table 4 showed that the most dominating impact categories were: Global warming, eutrophication, and acidification. In the following, environmental hot spots within these categories will be presented.

4.3 Environmental hot spots

In **Fig. 3**, **4** and **5**, environmental hot spots of the product chains of soybean meal (PO) and soybean meal (RSO) are shown. 'Energy' includes emissions (e.g., fossil CO_2) related to cultivation of soybeans, rape seeds, oil palms and spring barley, but also energy used on the milling plants. 'Fertilizer' includes processing (including energy) of artificial fertilizer used for the cultivation of soybeans, rape seeds, oils palms and spring barley. All three figures are dominated by large negative emissions from 'Avoided agricultural cultivation' which are saved emissions caused by 'avoided production of oil palms' (see Fig. 1) and 'avoided production of rapeseeds' (see Fig. 2), respectively. The avoided production of rapeseeds is largest, because the emissions per kg rapeseed are higher than the emissions from fresh fruit bunches (see Table 4). For all the figures, the positive contributions are smaller for soybean meal (PO) than for soybean meal (RSO). This is also clearly demonstrated in soybean loops, where an increased demand for soybean meal (PO) only results in production of 1,005 g soybean meal (and 12 g spring barley), whereas an increased demand for soybean meal (RSO) results in the production of 1,271 g soybean meal (and 117 g spring barley).

The contributions to **global warming potential** from different parts in the product chains of soybean meal (PO) and soybean meal (RSO) are shown in Fig. 3. The major contributor to global warming is the cultivation of soybean, where 8% of the greenhouse gases emitted during the soybean cultivation is fossil CO_2, and the rest N_2O. The N_2O comes from degradation of crop residues (e.g., straw) and biological nitrogen fixation. Contributions from 'freighter oceanic' and 'truck' are almost equal (the latter also includes avoided transportation of rapeseeds/palm kernels). For soybean meal (RSO), there is a considerable amount of avoided emission from fertilizer production. The demand for soybean meal (RSO) results in a shift from rapeseed cultivation (N fertilizer use = 167 kg N ha^{-1}) to soybean cultivation (N fertilizer use = 0 kg N ha^{-1}), thus saving fertilizer. The production of N fertilizer emits considerable more greenhouse gases, CO_2 and N_2O in particular, than P and K fertilizers. The contributions from 'truck' (transport of soybeans in Argentina (500 km)) and 'freighter oceanic' (shipping of soybean meal from Rosario in Argentina to Rotterdam in the Netherlands (12,082 km)) are very similar, despite the large difference in distance. This indicates that shipping is much more environmentally friendly than transport by truck.

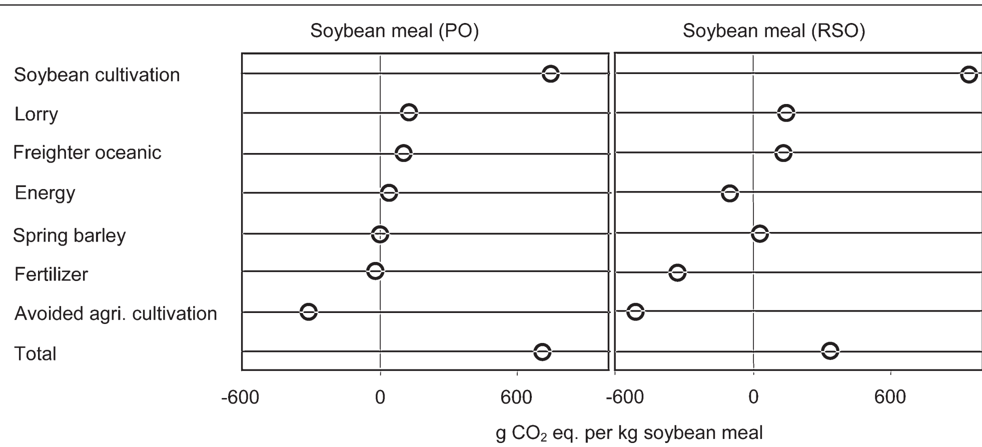

Fig. 3: Contribution to global warming potential from different parts of the product chain of soybean meal

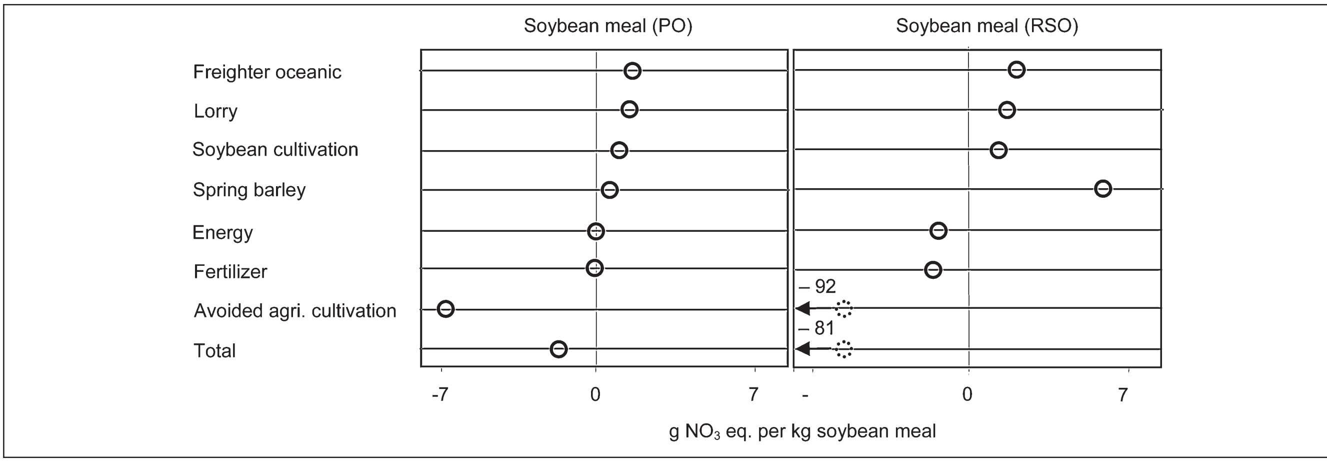

Fig. 4: Contribution to eurtrophication potential from different parts of the product chain of soybean meal

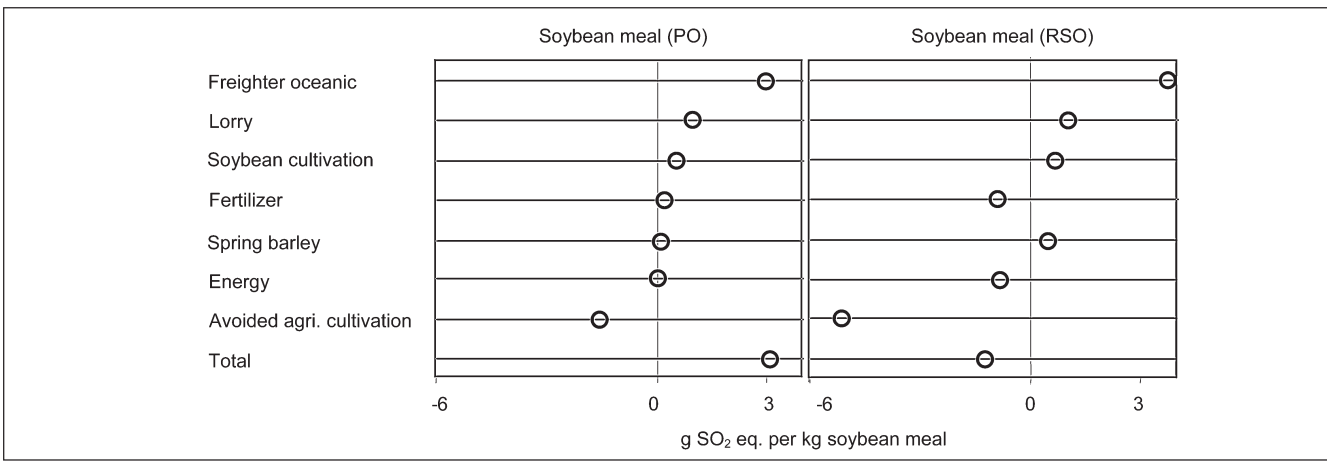

Fig. 5: Contribution to acidification potential from different parts of the product chain of soybean meal

Even though transport from Rotterdam Harbor to feedstuff companies and to farm gate was not part of the LCA in this study, we performed sensitivity analyses of this part of the product chain to see how much it could change the results. If the soybean meal (PO) was transported 650 km, this resulted in an increase of 20% in global warming potential.

Concerning **eutrophication potential** from soybean meal (RSO), it is clear that agricultural production avoided dominates the picture (see Fig. 4). This is because rapeseed contributes 139 g NO_3-eq. kg^{-1} produced compared with only 1 g NO_3-eq. kg^{-1} for soybeans (see Table 4). The environmental hot spot for eutrophication is 'spring barley' for soybean meal (RSO). The contribution from 'soybean cultivation' is smaller than the contribution from 'freighter oceanic' and 'truck'. Site-dependent impact assessment is not used in this study, so that, in the interpretation of the results, it must be taken into consideration that nutrifying substances emitted at sea damage vulnerable ecosystems (e.g., lakes, bogs) much less than on-land emissions.

The environmental hot spot regarding **acidification potential** (see Fig. 5) is 'freighter oceanic', but it must again be taken into consideration that acidifying substances emitted at sea are more harmless than if they were emitted on land. The second largest contributor is 'truck'. As for global warming, the production of N fertilizer avoided for rapeseed cultivation contributes negatively. When assuming 650 km of transport of soybean meal (PO) from Rotterdam Harbor, the acidification potential was increased by 32%.

4.4 Land use

As part of the inventory, data on land use for the different agricultural production systems were collected. The land used for the production of 1 kg of soybean meal (PO) and soybean meal (RSO) is 3.6 and 3.0 m^2year respectively (**Table 6**). The use of land in Argentina is higher than the total land use. The interpretation of this is of course that the use of 'one kg soybean meal (RSO)' costs 5.1 m^2year in Argentina, but saves 2.1 m^2year in Europe. Thus, the pressure on the pristine ecosystems in Argentina becomes quite obvious: Growing demands for soybean meal thus aggravate the pressures on land in other countries. At the end of the day, that eventually leads to a loss of biodiversity. Unfortunately, we have at present no method that reasonably translates the pressures on land into loss of biodiversity, although some efforts have been done to do so (Weidema & Lindeijer 2001, Lindeijer 2000, Mattson et al. 2000). These methodologies

Table 6: Land use per kg product (Unit: m²year). Data on soybean meals include both the soybean cultivation and the avoided productions of fresh fruit bunches and rape seeds

	Soybean meal (PO)	Soybean meal (RSO)	Soybeans	Fresh fruit bunches	Rapeseeds
Total	3.6	3.0	3.3	0.5	3.5
Of this in Argentina or Malaysia	4.0	5.1	3.3	0.5	0

can be criticized for not addressing a suitable way of linking known pressures on agricultural land in terms of occupation (m²year) with actual transformation between land use types. We argue that the most relevant aspect of land use impact is the transformation of pristine ecosystems into agricultural land. But we are only able to establish a link between the functional unit (1 kg soybean meal (RSO)) and the area occupied in Argentina (5.1 m²year) and Europe (–2.1 m²year). In addition to this, new research results linking landscape transformations and biodiversity using landscape ecology methodologies are being applied in Argentina to help to solve these limitations (Matteucci et al. 2004).

It should be noted that pesticide use was not included in the LCA results, but should be considered separately. The extensive use of broad-spectrum herbicides with glyphosate, fungicides and insecticides in the non-tillage RR soybean system may impact the health of farm workers and can have severe effects on biodiversity and aquatic environments, such as rivers and lakes, in the large areas where soybeans are virtually the only crop (see references in Ho & Ching 2003, Ho & Cummings 2005, Benbrook 2005). In the case where rapeseed is replaced by increased soybean production, this would increase the use of pesticides, also with potentially toxic effects on waterborne organisms but in a totally different location. The palm oil plantations also use glyphosate-containing herbicides. However, both the rapeseed and especially the palm oil cropping systems use significantly lower amounts of pesticides compared with the soybeans.

5 Discussion

5.1 Methodology: the use of consequential LCA

The rapeseed production had a large influence on the LCA of soybean meal (RSO) (see 'avoided agricultural cultivation' in Fig. 3, 4 and 5), and it even resulted in a negative eutrophication potential for soybean meal (RSO). This might appear irrational, but, according to the assumptions in these systems, when an extra amount of soybean meal is demanded, the vegetable oil production will shift from rapeseed oil to soybean oil. Seen in relation to eutrophication the rapeseed production is more harmful compared with the soybean production, because the nutrient surplus from the soybean production is low. Therefore, as long as an increased demand for soybean meal implies production of more soybean, at the same time it also induces a shift to an oil production causing less eutrophication. In the present LCA, it is assumed that an 'additional' production of 1 kg soybean oil results in a similar reduction in the production of rapeseed or palm oil (one-to-one substitution by kg oil). This is obviously a simplification because we assume (among other things) that prices remain unaffected and, thus, the model does not include price elasticity. However, this assumption is not only made in relation to system expansion, but in all the steps of a typical LCA inventory, as explained in Weidema (2003, p. 37).

Another obvious challenge of consequential LCA is the identification of affected processes, i.e., marginal processes/technologies. The present study shows significantly different results between the two soybean meal systems (PO vs. RSO), reflecting the use of palm oil or rapeseed oil as the marginal oil type. It cannot be established with certainty which of the oil types (palm or rapeseed oil) is the marginal oil – or whether it is a mix – or whether other types are included in this mix. The identification of the actual marginal is a great challenge, and an obvious source of uncertainty. However, it is not a better solution to assume that all plant oils are affected proportionally to their present market volume, which in reality would be the assumption behind an attributional LCA using average data.

In the present LCA of soybean meal, the changes in CO_2 emission caused by land-use changes (e.g., transformation of forest to cropland) were not included, due to conceptual and methodological limitations. Conceptually, it is debatable whether changes in above ground and soil carbon content due to changed land use should be included in an LCA of a product, especially when the functional unit is not related to carbon sequestration. If the impacts from 'land-use changes' related to crop cultivation are included in an LCA of agricultural products, this must be performed consequently. If, for example, the LCA data on soybean meal is used in an LCA of milk, the inclusion of the CO_2 emissions from land-use changes in Argentina should be combined with similar calculations of possible changed CO_2 sequestration in the dairy system (for example, more or less grassland versus maize in the crop rotation, which would influence soil organic matter). This is not presently done in LCAs involving agriculture for food nor bio-energy products (e.g., Cederberg & Flysjö 2004, Basset-Mens & van der Werf 2005, Heller et al. 2003, Kim & Dale 2005).

Methodological problems include the knowledge of land use before conversion, estimates of changes in above ground as well as below ground carbon content, both immediately and after initiation of cultivation and choice of depreciation time.

For example, the history of the area used is unknown, previously it might have been covered with crops, savannah, forest or something else. As there is a large difference in the amount of carbon stocked in these types of land (e.g., Fearnside 2000), it will influence the result strongly. Also the choice of depreciation time influences the result. Whether the emissions related to the land-use change should be ascribed completely to the crops cultivated during the first year or divided over the next 20–100 years of cultivation is debatable, and to determine the depreciation time demands better knowledge of the driving forces behind land-use changes. Despite the methodological limitations, we performed a sensitivity analysis, where it was assumed that the above-ground biomass of the forest before clear-cutting was 94 tons C ha^{-1} (area weighted mean for all tropical forests (Houghton 2005)), and the depreciation time was set at 20 year. Below-ground biomass and avoided deforestation related to palm oil production in Malaysia were not included in the calculation. According to the sensitivity analysis, the global warming potential inclusion of changes in CO_2 emissions caused by land-use changes increased the global warming potential dramatically from 721 g to 5.7 kg CO_2 eq. per kg soybean meal (PO).

In consequential LCAs, the use of marginal data is to be preferred, because the consequential LCA seeks to reflect the environmental consequences of an increasing demand for a certain product. But when are data marginal and when are they average? Argentina is the marginal soybean meal producer as argued in the 'system delimitation' section and, therefore, we have used averaged data on soybean meal yields in Argentina. Preferably, we should have used yield data from the marginal soybean producers within Argentina. Marginal data on yields of soybeans and fresh fruit bunches were not available and, as we did not find a reason to believe that the marginal yields would be very distinct from the average yields, we used average data.

N_2O emissions from cultivation of soybeans, rapeseeds, spring barley and oil palms appeared to have a large impact on the global warming potential per kg soybean meal. This is in good agreement with other studies, showing that N_2O plays a major role in the greenhouse gas emissions from agricultural production (Olesen et al. 2006; Dalgaard et al. 2006). N_2O emissions from soybeans, rapeseeds and spring barley were calculated according to the IPCC guidelines. However, we had difficulties in finding literature or methods for estimating N_2O emissions from oil palms, and we therefore used the same data as for soybeans, but adding N_2O emitted from the N fertilizer applied to the oil palms. This was unsatisfactory as the N_2O turned out to be important for the final result of the LCA of soybean meal (PO).

5.2 Comparison with previous studies

A majority of the previous LCA studies on livestock products, where soybean meal was included, does not directly present LCA data on soybean meal. However, Eriksson et al. (2004), who based the soybean inventory on data from Cederberg & Darelius (2001), have published LCA data on soybean meal. Ecoinvent Centre (2004) also provide data for the LCA database on 'soybean scrap', based on soybeans produced in Switzerland. Economic allocations were performed in both the above-mentioned studies. The environmental impacts of producing one kg of soybean meal, according to Eriksson et al. (2004) and Ecoinvent Centre (2004), are as follows: Global warming: 730 and 507 g CO_2 eq.; acidification 8 and 13 g SO_2 eq.; eutrophication: 541 and 198 g NO_3 eq. (LCIA method applied for Ecoinvent data: EDIP97 (Wenzel et al. 1997, updated version 2.3)). The results on global warming are in good agreement with ours (soybean meal (PO) in Table 4 and soybean meal (69%) in Table 5), whereas our results on acidification and eutrophication are considerably lower. These differences are not only due to the use of consequential versus attributional LCA, but to a larger extent due to the estimated emissions. For example, the soybean cultivation in Ecoinvent contributes negatively to GWP, because biotic fixation of CO_2 from the atmosphere is considered as a negative contribution to global warming potential. In our calculation, we consider the biotic fixated CO_2 as neutral, because it will be released to the atmosphere after digestion by livestock. The similarity of the results on global warming potential must be ascribed to accidental occurrence. The environmental hot spots in Ecoinvent Centre (2004) are, as in our study, transport by freighter oceanic and truck. But, in our results, site-dependent aspects are not taken into consideration.

Cederberg & Flysjö (2004) estimated that nitrate leached from soybean cultivation in Cerrado in Brazil equaled 36 kg N ha^{-1}, whereas we estimated no nitrate leaching. In the study of Cederberg & Flysjö (2004), the input of N (fertilizer and BNF) to the soybean cultivation was assumed to be 230 kg N ha^{-1}. We find this is a very high estimate because the average N application (fertilizer) to soybean fields in the Pampas was only 2 kg N ha^{-1} in 2002 (FAO 2004). According to Austin et al. (2006), nitrate was not leached from the soybean fields in the Pampa region. In contrast, a substantial net loss of nitrogen at the regional scale was taking place, and the current agricultural practices in the Pampa region are essentially 'mining', the nutrient capital of the region (Austin et al. 2006). Unfortunately, this export of nutrients out of the region probably leading to nutrient deficiencies in the soil, is not captured in our LCA of soybean meal. The discrepancy between the results of Cederberg & Flysjö (2004) and our results might be due to differences in cultivation practices in Brazil and Argentina. For example, the fertilizer use efficiency in Argentina is four times higher than in Brazil (Austin et al. 2006), and the fertilizer use is generally considerably lower.

5.3 Relative impact of the Danish soybean consumption: Scaling up from FU to national level

The results demonstrate that the soybean meal consumption in Europe has an impact on the global environment (e.g., global warming) and on the local environment outside Europe (e.g., acidification, land use). But what is the magnitude of these environmental impacts from soybean meal production compared with the environmental impacts of the livestock production itself? As an example, we compared greenhouse gas emissions: According to Gyldenkærne & Mikkelsen (2004), 10.5 million tonnes CO_2 eq. were emitted from the entire Danish agricultural sector in 2002. In the same year, 1.5 million tonnes of soybean meal were imported to Denmark (Statistikbanken 2006). If the transport from Rotterdam Harbor to a Danish feed company is set at 650 km, the greenhouse gas emission is 869 g CO_2 eq. per kg soybean meal (PO) (=721 * 1.20). This results in a 'soybean meal related greenhouse gas emission' of 1.3 million tonnes CO_2 eq., which is approximately equivalent to 12% of the greenhouse gas emitted directly from the Danish agricultural sector. So, in addition to the 10.5 million tonnes emitted from the agricultural sector, an extra amount of 1.3 million tonnes is emitted as a consequence of the soybean meal import to Denmark. Unfortunately, it was not possible to estimate the environmental impact from pesticide use and loss of biodiversity caused by pressure on pristine ecosystem.

6 Conclusions

Consequential LCA was successfully performed on soybean meal. An increased demand for soybean meal implies an increased production of soybean oil as both commodities originate from the soybean. This soybean oil will substitute the marginal vegetable oil on the market and, therefore, an increased demand for soybean meal results in an avoidance of the production of the crop producing the marginal oil. This avoided crop production (and other affected crops) was included in the calculations. A recent study by Schmidt and Weidema (2007) has identified palm oil as the marginal oil. However, a shift to rapeseed oil might be possible, as the two vegetable oils are comparable in many aspects. To be prepared for such a shift and to analyze to what extent the choice of marginal oil affects the result of the LCA, two LCAs on soybean meal were performed: One with palm oil as the marginal oil (soybean meal (PO)) and one with rapeseed oil as the marginal oil (soybean meal (RSO)).

The functional unit was 'one kg soybean meal produced in Argentina and delivered to Rotterdam Harbor'. The characterized results from the LCA on soybean meal (PO) were 721 g CO_2 eq. for global warming potential, 0.3 mg CFC11 eq. for ozone depletion potential, 3.1 g SO_2 eq. for acidification potential, –2 g NO_3 eq. for eutrophication potential and 0.4 g ethene eq. for photochemical smog potential. The potential environmental impacts (except photochemical smog) were lower for soybean meal (RSO), because the avoided environmental impact was larger from rapeseed compared with oil palms.

Normalized results showed that the most dominating impact categories were: global warming, eutrophication and acidification. The 'hot spot' in relation to global warming was 'soybean cultivation', dominated by N_2O emissions from degradation of crop residues (e.g., straw) and during the biological nitrogen fixation. Eutrophication is not a major problem in the soybean cultivation. In relation to acidification, the transport of soybeans by truck was important, and sensitivity analyses showed that the acidification potential was very sensitive to increased transport distance by truck.

7 Recommendations and Perspectives

This study clearly shows that consequential LCAs are quite easy to handle and that LCA data on soybean meal are now available for consequential (or attributional) LCAs on livestock products. But there are, of course, some limitations to this analysis. First of all, it is important to know which of the related product systems, e.g., the vegetable oil system, are marginal. We would appreciate it if International the Journal of Life Cycle Assessment had articles on the developments on, for example, marginal protein, marginal vegetable oil, marginal electricity (related to relevant markets), marginal heat, marginal cereals and, likewise, on metals and other basic commodities.

It is also recommended that more effort be put into describing the impacts of land use. With a growing global population and increasing demands on meat instead of vegetable products, the pressures on arable land and eventually reclamation of natural habitats for farming puts tremendous pressures on the natural habitats in many places around the world. It is thus pivotal that we become able to manage loss of biodiversity as a fundamental impact category in LCA studies.

Soybean expansion in Latin America represents a powerful threat to biodiversity in Brazil (Cerrados), Argentina (Chaco, Yungas and Monte Ecoregions), Paraguay and Bolivia. In addition to herbicide use and genetic pollution, the massive requirement for infrastructure projects (highways, ports and railways) are also threats to the high biodiversity that presently exists in Latin America (Alteri and Pengue 2006).

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Environmental assessment of Danish pork

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Environmental assessment of Danish pork

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Abstract

Background, Aim and Scope. The pig production in Denmark and other European countries affects the environment in several ways. The pork production chain is complex, and there still is a need to identify the most polluting parts of the product chain. The objectives of the study were therefore:

- To provide LCA data on Danish pork and demonstrate an application of consequential LCA modelling within food production
- To develop the use of consequential LCA methodology for agricultural products using representative farm account data and an economic assessment method
- To identify the processes in the product chain of pork with the largest environmental impact (hot spots)

The functional unit was one kg of pork produced in Denmark and delivered to Harwich Harbour in Great Britain.

Materials and Methods. Consequential LCA modelling was used to show the consequences of producing one additional kilogram of Danish pork and transporting it to Harwich Harbour. Identification of the processes and products that would be affected by a change in demand for pork was facilitated by already established data sources: Agricultural data used in the study were primarily from a national agricultural model based on farm types that are representative of the entire Danish agricultural sector. The econometric sector model ESMERALDA was

used to identify the farm types affected by a change in demand for pigs and crops. For the Impact Assessment, EDIP 1997 (version 2.03) was used. The environmental impact categories considered were global warming, eutrophication, acidification and photochemical smog, and in addition land use (unit: m^2year) was calculated.

Results. The environmental impacts were 3.77 kg CO_2 eq. global warming potential, 319 g NO_3 eq. eutrophication potential, 59 g SO_2 eq. acidification potential, and 1.27 g ethene eq. photochemical smog potential per kg Danish pork delivered to Harwich Harbour. The area occupied per kg pork was 9.1 m^2year. Farms for fattening pigs (size: 30-100 kg) and weaners (size: 7-30 kg) were identified as being the environmental hot spots for both global warming, eutrophication and acidification. Within the farms, nitrate, ammonia and nitrous oxide emissions were the major contributors to eutrophication, acidification and global warming, respectively. The transport of the pork by lorry and oceanic freighter to Harwich Harbour contributed less than 2% for all impact categories, except for photochemical smog where the contribution was 5%.

Discussion. The results were, in general, in accordance with previously published studies, although consequential modelling was not used in these. Sensitivity analyses showed that even if the Danish pork was transported to Japan, which is the third largest importer of Danish pork, the global warming potential per kg Danish pork (delivered to Tokyo Harbour) would increase by only 5%.

Conclusions. It proved possible to generate LCA data using consequential modelling for an agricultural product based on representative farm data and to identify affected farm types using the econometric sector model ESMERALDA. Main contributors to all impact categories, except photochemical smog, were the farms producing fattening pigs and weaners. Particularly, the nitrogen-containing compounds nitrate, nitrous oxide, and ammonia were responsible for much of the environmental impact. Efficient use of nitrogen on pig farms is therefore an important issue if the environmental impact from pig production is to be reduced.

Perspectives. The distance food is transported (food miles) was revealed not to be very important for any of the impact categories. Instead we recommend that the nitrogen use efficiency on pig farms is focused on in order to improve the environmental impact of Danish pork. There are several sources of greenhouse gases in the product chain of pork and most of these will be reduced if the nitrogen use efficiency is improved at farm level.

Keywords: Agriculture; consequential LCA; LCA; pork; pig; system expansion

1 Introduction

Livestock keeping is responsible for 18 percent of the greenhouse gas emissions and 64 percent of the anthropogenic ammonia emissions in the world (Steinfeld et al. 2006). However, future population growth and an increase in consumption will raise the demand for meat (Elfering & Nonhebel 2006), and, specifically, an increase in pork production and trade is forecasted for the next decade (FAPRI 2006). Therefore, more than ever, it is pertinent to get precise information on the environmental load from different livestock products in order to identify environmental hot spots and reduce emissions. In Europe, pork production is also expected to increase, but only slightly due to strict environmental regulations and animal welfare requirements (FAPRI 2006). In 2002-2004, 65% of the European pigs produced came from Germany, France, Spain and Denmark.

The debate on the impact of pork production on the environment has increased in Europe, not only because the amount produced is continually increasing, but also because pork production becomes increasingly concentrated in specific geographical areas. According to OECD (2003), there is a general tendency within all OECD countries for pig production to become more intensive, with an increase in the average number of animals per unit land area and per pig farm. The concern is mostly directed towards the local environment (e.g. lakes, fens and inlets), which may be affected by – for example - nitrate or phosphate leached from the fields of adjacent pig farms or ammonia evaporated from their slurry tanks (European Environment Agency 2005).

The increasing global character of the pork chain with much of the concentrates being imported to Europe from South America (soybeans) and Asia (palm oil) necessitates an assessment of the environmental impact on a global scale including concentrate production. It is claimed that the soybean production escalates the use of genetically modified plants, the use of the pesticide glyphosate (Pengue 2006; Benbrook 2005), greenhouse gas emissions (Dalgaard et al. submitted), and the loss of biodiversity and deforestation (Benbrook 2005; Steinfeld et al. 2006).

Moreover, the increased transport arising from globalization of food chains impacts the environment negatively. Thus the term ‘food mile’, which is a measure of the transport distance of a food product from producer to consumer, has become part of the debate (Pretty et al. 2005; Saunders et al. 2006)

The product chain of pork is complex, and so is the environmental assessment of pig and pork products. The LCA methodology has previously been used for environmental assessment of pig and pork (Eriksson et al., 2005; Cederberg & Flysjo 2004; Carlsson-Kanyama 1998; Basset-Mens & van der Werf 2005; Halberg et al. submitted). In most of these studies the attributional LCA approach was applied. Only Halberg et al. (submitted) applied the consequential LCA approach.

There has been an ongoing discussion whether to use consequential or attributional LCA (Heijungs & Guinée, submitted; Tillman, 2000; Ekvall and Finnveden, 2001; Curran et al., 2005; Ekvall and Andræ, 2005; Cederberg & Stadig, 2004). In attributional LCA, only material and energy flows physically linked to the considered product are taken into account, whereas in consequential LCA products outside the (‘physical’) life cycle are also included if these products are affected by a change in demand. A fundamental question in consequential LCA is who and what will be affected by a change in demand for the product in question. For example, a change in demand for pork implies a change in demand for fattening pigs, weaners, concentrates, electricity, heating oil, fertilizer, etc. A large challenge in consequential LCA is to define who the suppliers of these items are and what technology or type of production is used. Besides providing LCA data on pork and identifying environmental hot spots, the aim of this article also is to show how affected suppliers can be identified and how this, in some cases, can be facilitated by using already existing economic models and data.

2 Goal and scope definition

The objectives of the study were:

- To provide LCA data on Danish pork and demonstrate an application of consequential LCA modelling within food production.
- To develop the use of consequential LCA methodology for agricultural products using representative farm account data and an econometric sector model.

- To identify the processes in the product chain of pork with the largest environmental impact (hot spots).

The functional unit is defined as 1 kg Danish pork (carcass weight) delivered to Harwich Harbour in Great Britain.

Consequential LCA modelling was used. This implies that the processes and their technologies involved are those actually affected by the *additional* production of 1 kg pork, as opposed to the average kg produced. The affected processes are identified as those most sensitive to a change in demand. In a market with increasing output over time, the producers that can increase their output *most efficiently* (at the lowest costs) will react more strongly to an increase in demand than less efficient producers or producers that are constrained with respect to one or more of their production factors. In a market with decreasing output over time, the *least efficient* producers with excess capacity will typically be most sensitive to a change in demand. Processes affected by a change in demand for pigs are presented in figure 1. The lowest lefthand corner of the figure shows 'Danish pork delivered to Harwich Harbour' . The pork is transported by freighter oceanic and lorry from the slaughterhouse, and the pigs slaughtered at the slaughterhouse are primarily fattening pigs produced at specialized fattening pig farms, but also culled sows from specialized weaner farms. The fattening pig farm purchases weaners (30 kg) from the specialized weaner farm.

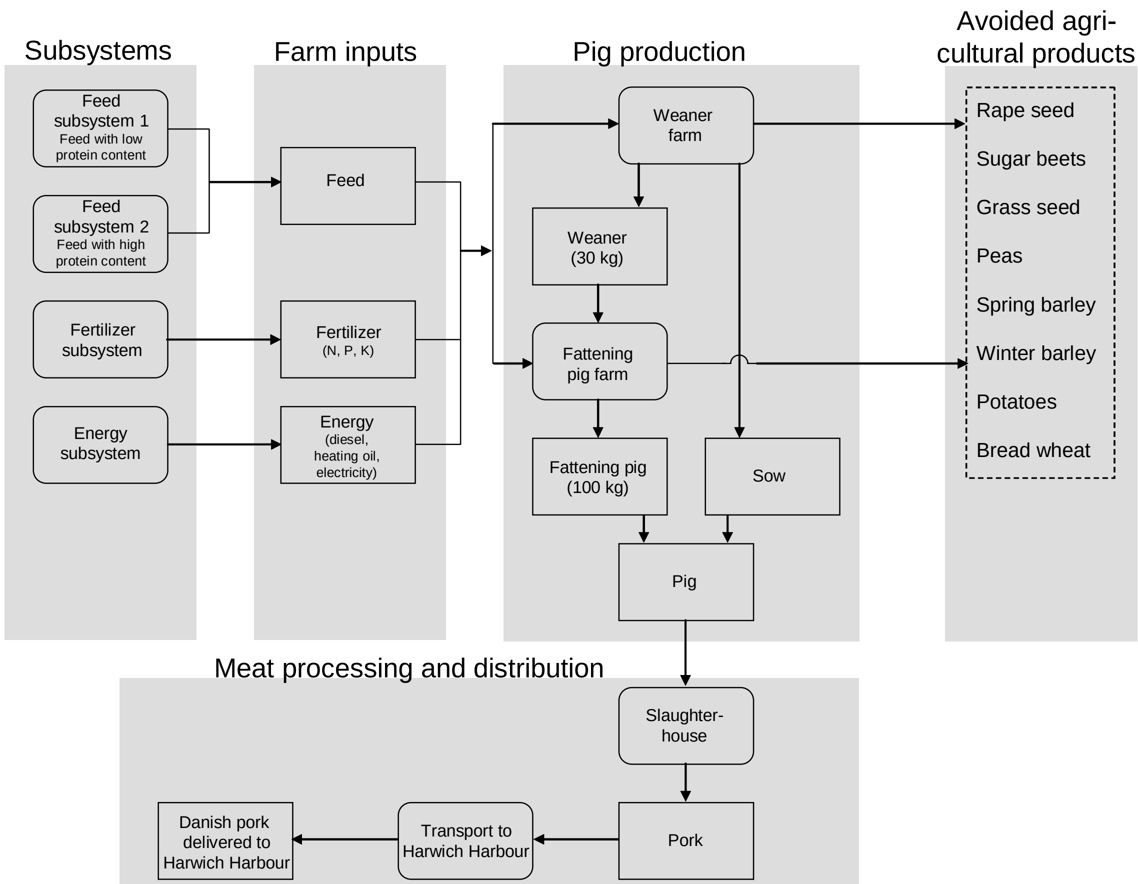

Fig. 1. The product chain of Danish pork. Boxes with angular corners represent products and boxes with rounded corners represent processes.

Figure 1 also illustrates the 'farm inputs', which are the commodities used directly on the farm, and 'subsystems', which are the processes outside the product chain of pig that are affected by the increased demand of the farm inputs. Inputs to the two farms are feed, fertilizer and energy.

The two pig farms also produced cash crops (rape seed, sugar beets, etc.), as shown to the right in figure 1 under the heading 'avoided agricultural products'. According to the concept of system expansion used in consequential LCA, each of these products will replace similar products, or products with the same function, on the market (Weidema 2003). Therefore, the environmental impact from the product chain of pig was only to be ascribed to pig, no allocation was needed. The environmental burdens due to replaced production of bread wheat, rape seed, sugar beets etc. were subtracted from the environmental impact of pig. The EDIP 1997 (Wenzel et al. 1997, updated version 2.3) was used for the impact assessment, and the following impact categories were included: global warming, eutrophication, acidification and photochemical smog. Results for land use per kg pig in terms of land occupation (unit: m^2year) are also presented.

3 Inventory

3.1 Data on fattening pigs, weaners and cash crops

Data on the agricultural products shown in figure 1 (e.g. weaners, fattening pigs, rape seed, sugar beets) derive from a national agricultural model established by Dalgaard et al. (2006) on the basis of farm account data, which are representative for the Danish agricultural sector. The national agricultural model contains data on resource use, production and emissions from 31 typical farm types. These 31 farm types are representative of the Danish Agricultural sector (in the year 1999). Within each farm type there is coherency between resource use (e.g. feed, fertilizer), production (e.g. pig, cash crops) and emissions. For a more comprehensive description of the 31 representative farm types, see Dalgaard et al. (2006).

The national agricultural model contains six specialized pig farm types that represent 76% of the fattening pig production in 1999. The rest of the pigs are produced on different types of farm, such as part-time farms, dairy farms or arable farms. The different kinds of cash crops are not only produced on the arable farms, but also on pig and milk producing farms. But which farm types are affected by a change in demand for fattening pig or wheat? An answer to the question can be obtained using the econometric sector model ESMERALDA (Jensen et al. 2001) as explained in the following. The identification of the affected fattening pig supplier is used as an example, but the same procedure was followed to identify the affected crop suppliers.

The observed structure of production across farm types is assumed to be an equilibrium situation in the sense that it is compatible with the economic framework conditions in the corresponding period. Now assume a marginal shift in the demand for pork, see figure 2. Assuming market equilibrium, this demand shift will affect prices, which in turn will affect pork supply (and possibly the supply of other agricultural products) possibly on several of the farm types. These supply responses were determined by means of the ESMERALDA model (Jensen et al., 2001), which comprises econometrically estimated behavioural parameters for eight broad farm categories: full-time pig, cattle and crop farms, as well as part-time farms, on clay and sandy soil. Parameters were estimated from farm-level FADN data for the period 1973/74 to 1995/96. The eight broad farm categories can to a large extent be considered as an aggregation of the 31 farm types, and hence behavioural parameters (e.g. elasticity of product transformation, elasticity of input substitution, parameters of marginal factor productivity) for one farm category were assumed to represent all farm types within this category. For example, behavioural parameters for the farm category of pig farms on sandy soil were assumed to represent all three pig farm types on sandy soil in the national agricultural model. Together with structural information (e.g land allocation, livestock density, composition of livestock production, capital intensity, cost structure) on each farm type, these behavioural parameters determine the marginal output response for each farm type. While all farm types react slightly to a change in demand, we only used the farm type with the largest total marginal response to pork supply. The marginal producers of cash crops and feed grain were identified by the same procedure.

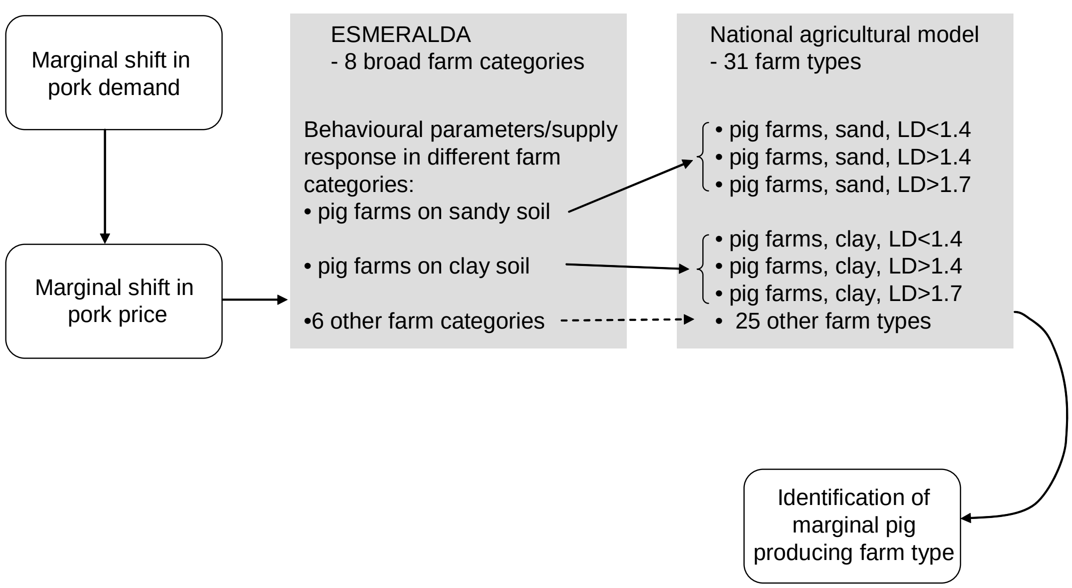

Fig. 2. Procedure for identification of the marginal pork producing farm type amongst the 31 farm types from the national agricultural model

The marginal pork producer type was hence characterized as a specialized pig farm located on sandy soil, with a livestock density (LD) below 1.4 livestock units per hectare (one livestock unit equals 36 fattening pigs (30-100 kg)). This farm type represented 2310 pig farms and 11 % of the total pig production in the baseline situation, but around one third of the marginal increase in pig production. That the marginal fattening pig is produced on a specialized pig farm on sandy soil seems reasonable, because the number of specialized pig farms is increasing and the pig farms are moving to the west of Denmark where sandy soil dominates (Dalgaard et al. 2006). The identified farm type represented both weaner farms and fattening pig farms, and in order to establish a farm type producing exclusively fattening pigs, the farm accounts were subdivided into three new farm types: one representing pig farms producing only fattening pigs, one representing pig farms producing only weaners (and culled sows), and one group producing both types of pigs. The farm type producing only fattening pigs was selected as data source for the fattening pig farm and represented 1195 fattening pig farms in Denmark, corresponding to 8.2% of the total fattening pig production in 1999. The farm type producing only weaners was selected as data source for the weaner farm and represented 213 weaner farms in Denmark, corresponding to 2.6% of the total weaner production in Denmark in 1999.

The characteristics of the two farm types are presented in table 1, and the inventory for the two farm types is presented in table 2. Data in table 1 and 2 form part of the national agricultural model established by Dalgaard et al. (2006). The majority of the agricultural area at the two farm types is under cereal production, as shown in table 1. Cereal grown at the farms is used as feed at the farms, except for 65% of the wheat, which is sold as wheat for bread-baking (table 2). Less than 20% of the agricultural area on the two farms is used for growing rape seed and other cash crops (sugar beets, peas, potatoes and seeds). Self-sufficiency in feed supply (measured in N) is lowest for the weaner farm, because the feed demand per ha for the weaner farm is higher, while the production of feed grain is almost the same on the two farms. Table 2 shows that the amount of feed grain and soybean meal imported by the farms is highest for the weaner farm. On a yearly basis, the two farms produce 1402 fattening pigs (size 100 kg) and 2717 weaners (size: 30 kg), respectively. N excretion per animal (table 1) is from Poulsen et al. (2001). The N and P surpluses are the annual imports of N and P to the farms that are not exported in products. The N and P surpluses are calculated following the procedures of Halberg et al. (1995) and Kristensen et al. (2005). For further details see Dalgaard et al. (2006).

Table 1. Characteristics of fattening pig farm and weaner farm. Data source: Sub types of farm type 20 in Dalgaard et al. (2006)

Characteristics	**Unit**	**Fattening pig farm**	**Weaner farm**
Agricultural area	ha	71	73
Grain	%	68	68
Rape seed	%	10	8
Other cash crops	%	8	11
Grass and set-aside	%	14	13
Self sufficiency in feed (measured in N)	%	41	35
Livestock density	LU ha^{-1}	0.8	0.8
Fattening pigs produced	Animals $year^{-1}$	1402	0
Weaners produced	Animals $year^{-1}$	0	2820
N excretion			
Weaners (7-30 kg)	kg N $animal^{-1}$		0.64
Fattening pigs (30-100 kg)	kg N $animal^{-1}$	3.15	
N surplus	kg N ha^{-1} $year^{-1}$	95	107
P surplus	kg P ha^{-1} $year^{-1}$	13	16

Table 2. Inventories of the fattening pig farm type and the weaner farm type, showing farm inputs, sold cash crops and emissions per year. Data source: Sub types of farm type 20 in Dalgaard et al. (2006)

Farm inputs	**Unit**	**Fattening pig farm**	**Weaner farm**
Feed grain	kg ha^{-1}	268	668
Soybean meal	kg ha^{-1}	791	859
Mineral feed	kg P ha^{-1}	9	14
Weaners (size: 30 kg)	kg ha^{-1}	559	0
Fertilizer, N	kg ha^{-1}	92	90
Fertilizer, P	kg ha^{-1}	10	7
Fertilizer, K	kg ha^{-1}	41	36
Electricity	kWh ha^{-1}	385	559
Heating oil	MJ ha^{-1}	471	2366
Diesel	MJ ha^{-1}	3878	3912
Lubricant oil	liter ha^{-1}	11	11
Sold cash crops			
Bread wheat	kg ha^{-1}	1141	1147
Rape seed	kg ha^{-1}	257	241
Sugar beets	kg ha^{-1}	689	867
Straw	kg ha^{-1}	384	437
Peas	kg ha^{-1}	84	131
Grass seeds	kg ha^{-1}	26	50
Emissions			
Methane	kg CH_4 ha^{-1}	19	16
Ammonia	kg NH_3 ha^{-1}	30.4	35.7
Nitrous oxide	kg N_2O ha^{-1}	6.3	7.0
CO_2-fossil	kg CO_2 ha^{-1}	337	340
Nitrate	kg NO_3 ha^{-1}	307	325
Phosphate	kg HPO_4 ha^{-1}	1.2	1.6

Annual purchased farm inputs by the two pig farms are shown in table 2. The weaner farm purchased more feed grain, soybean meal and mineral feed compared with the fattening pig farm, while the amounts of fertilizer imported are almost equal for the two farms. More electricity and heating oil are used on the weaner farm, as it uses a climate-controlled livestock building for piglets. The tractor diesel consumption is the same for the two farms, as almost the same crops are grown on the farms (see table 1). The methane and nitrous oxide emissions presented in table 2 are calculated according to guidelines from the Intergovernmental Panel on Climate Change (IPCC 1997; 2000), while ammonia emission from animal houses, manure handlings and fields were calculated using standard values from Hutchings et al. (2001), and nitrate and phosphate emissions were calculated using the values for N and P surpluses presented in table 1, but taking into account that not all N and P surpluses are lost as nitrate or phosphate. For further detail, see Dalgaard et al. (2006).

3.2 Farm input data and identification of subsystems affected by a change in demand

Feed, fertilizer and energy inputs are imported by the two pig farms (figure 1). An increase in consumption of these products does not automatically lead to the production of identical products but might in many cases lead to a product substitution, where products outside the ‘physical’ product chain of the pig are affected by the increased demand for farm inputs. These product substitutions will be explained in the following.

3.2.1 Feed

An increased production of fattening pigs implies an increased demand for feed. Typically the purchased feed mixtures used on Danish pig farms contain several ingredients (e.g. rape seed meal, soybean meal, wheat, barley, fish meal, sunflower meal), but using the consequential approach it is sufficient to have LCA data on the crops, which are affected by an increased demand for feed.

When a fattening pig farm purchases – for example - rape seed meal, there will be an increased demand for rape seed meal on the market. But because the most competitive protein meal on the world market is soybean meal, and not rape seed meal as argued by Dalgaard et al. (submitted) and Weidema (2003), the increased demand for rape seed meal will result in an increased production of soybean meal. Consequently, it is necessary to have LCA data on soybean meal, but not on rape seed meal and other ‘not-affected crops’. Because soybean meal has too high a protein content to meet both the protein and energy demand of pigs, also grain production is affected by an increased demand for feed. So even though several feed ingredients are purchased by a pig farm, only LCA data on soybean meal and grain are needed. The amounts of soybean meal and grain are calculated on the basis of energy and protein content of the feed ingredients needed to satisfy the protein and energy requirements of the pigs. In the following, the data sources on soybean meal and grain will be explained.

LCA data on soybean meal produced in Argentina and delivered to the Netherlands are used (Dalgaard et al. submitted), and LCA data on transport of the soybean meal to Denmark is added. In the establishment of LCA data on soybean meal it is assumed that the co-product soybean oil displaces palm oil cultivated in Asia (Dalgaard et al. submitted; Schmidt & Weidema, submitted). Change in the demand for grain may affect the production of feed grain in many locations including Denmark. In our calculations we have used a similar approach to identify the affected producers in Denmark as described for the pig farm types. Consequently, LCA data on feed grain are (similar to the fattening pig and weaner data) from the national agricultural model. The feed grain is a mix of spring barley (25%), winter barley (25%) and wheat (50%). See Dalgaard et al. (2006) for details on modelling of emissions and resource use, and see www.LCAfood.dk.

3.2.2 Fertilizer

Nitrogen, phosphorus and potassium artificial fertilizers are used on the two pig farms and for production of the feed purchased by the two pig farms. A change in demand for artificial fertilizer affects the less competitive fertilizer producers, as the European market has experienced a decrease in the consumption of fertilizer due to environmental restrictions (Weidema 2003, p. 73). LCA data on production of artificial fertilizer represent the least competitive fertilizer producers and are from Patyk & Reinhardt (1997).

3.2.3 Energy, animal housing and equipment

Diesel purchased by the farms is used for the agricultural machinery, and these data are based on Borken et al. (1999), but moderated to average load (for further information, go to www.LCAfood.dk). Data on electricity (electricity from fuel gas power plant in the Netherlands), transport by lorry (16t), and heating oil used for animal housing are all from the Ecoinvent Centre (2004). Data on the manufacturing and transport of materials used for animal housing and equipment (e.g. concrete, wood) are from Halberg et al. (submitted), who used LCA data on the materials from the Ecoinvent Centre (2004).

3.2.4 Avoided products

No allocation between co-products has been performed. Co-product outputs displace other products, which are thus included as avoided products by system expansion. Emissions from the avoided products (see figure 1) are subtracted from the emissions from the farm. LCA data for the avoided production have the same origin (Dalgaard et al. 2006) as LCA data on feed grain. See the description in section 3.1.2.1.

3.2.5 Data on meat processing and transport

Fattening pigs and sows from the farms were assumed to be transported by lorry (size:16 tonnes) 50 km to the slaughterhouse. Data on the slaughtering process were based on Pontoppidan & Hansen (2001) and represented more than 90% of the pig slaughtering in Denmark in 1997/1998. The resource use per 100 kg fattening pig or sow was 8.4 kWh electricity, 13.3 kWh heat and 200 litres of water. The assumed carcass weight was 76.3% of the living weight (Poulsen et al. 2001), which is in good agreement with the results of Andersen et al. (1999). Fifteen kilo waste in terms of bone, blood and meat meal was produced per 100 kg living weight fattening pig or sow. This was either transported to an incineration plant where it was used for energy production, used as feed or destructed. The possibly avoided production of feed caused by reuse of bone, blood and meat meal was small and was therefore excluded from the calculations. For further details see www.LCAfood.dk.

Transport by lorry (size 16 tonnes) from the slaughterhouse to the freighter oceanic was assumed to be 126 km, which equals the distance from Horsens Slaugtherhouse (largest slaughterhouse in Denmark), to Esbjerg Harbour (largest pork exporting harbour in Denmark). LCA data on transport by freighter oceanic are from Ecoinvent Centre (2004). The distance from Esbjerg Harbour to Harwich Harbour is 619 km.

4 Results

Characterization and normalization results are presented in this section. Afterwards environmental hot spots and land occupation are described.

4.1 Characterized and normalised results

The characterized and normalised results for 1 kg pork delivered to Harwich Harbour are presented in table 3. The normalization showed that the pork production contributed more to global warming, eutrophication, acidification than to the other impact categories.

Table 3. Impact potentials. Normalised and characterized results. Functional unit: 1 kg pork delivered to Harwich Harbour

Impact category	Characterization	Normalization
Global warming	3.77 kg CO_2 eq.	4.3E-04
Eutrophication	319 g NO_3 eq.	1.1E-03
Acidification	59 g SO_2 eq.	4.7E-04
Photochemical smog	1.27 g ethene eq.	6.3E-05

Table 4 shows the potential contributions from different parts of the product chain of fattening pigs. The respective parts of the product chain of pork are defined according to inputs to the fattening pig farm ('fertilizer', 'feed', 'energy' and 'weaners') and the 'agricultural productions' displaced by the outputs from the fattening pig farm. The avoided agricultural products are: bread wheat, rape seeds, sugar beets, peas and grass seeds (see table 2). 'Energy' includes oil for heat, diesel refining and distribution, lubricant oil used for machinery, and electricity used on the fattening pig farm. Energy used for production of weaners, feed and fertilizer is included in 'weaner farm', 'feed' and 'fertilizer' respectively. Production of the inputs to the fattening pig farm also encompasses co-productions (e.g. rape seed from the weaner production, soybean oil from soybean meal production). The avoided productions are included under their respective inputs to the fattening pig farm, and cannot directly be seen from table 4. 'Fertilizer' includes artificially produced N, P and K fertilizer. 'Feed' includes solely the feed purchased by the fattening pig farm. The contribution from animal housing and equipment is less than one percent for all impact categories and thus omitted from table 4.

In the following, the environmental hot spots for each of the impact categories are presented. All percentages refer to equivalents within the respective impact categories (see table 4).

Table 4. Environmental hot spots in the product chain of pork based on characterized results. Values are percentages of the total emission from the considered part of the product chain. Functional Unit: One kg Danish pork (slaughtered weight) delivered to Harwich Harbour in Great Britain. Calculated according to EDIP method (Wenzel et al. 1997)

Part of product chain of Danish pork	**Global warming**	**Eutrophication**	**Acidification**	**Photochemical smog**
Inputs to fattening pig farm				
Fertilizer	13	1	3	2
Feed	14	2	6	29
Energy	5	0	1	20
Weaners	23	22	26	17
Fattening pig farm	40	71	60	22
Avoided agricultural production	-21	-22	-12	-14
Slaughterhouse	4	4	3	5
Transport to UK	1	0	1	5

Global warming potential: Main contributors to the global warming potential are the fattening pig farm (40%) and the weaner farm (23%). Of the greenhouse gases emitted on the fattening pig farm 72% is nitrous oxide (fig. 3), whereas fossil CO_2 from the use of agricultural machinery only contributes 11%, and methane from slurry handling contributes 17%. The different sources of nitrous oxide emissions at the fattening pig farm are presented in fig. 4, and it shows that 43% of the nitrous oxide comes from denitrification of nitrate leached from the field, and 44% originates from denitrification of N fertilizer (artificial fertilizer and slurry) applied to the fields. Table 4 shows that production and distribution of fertilizer imported to the fattening pig farm contributes 13% of the greenhouse gases emitted. Of this, 94% is related to the production of N fertilizer and only 6% to the production of P and K (not shown), because these fertilizer types are used in smaller amounts and less energy is used for their manufacture. Of the emitted greenhouse gases, 14% is are related to the production of feeds imported to the fattening pig farm, and 79% of this can be ascribed to production and distribution of soybean meal. So despite the fattening farm and weaner production, production and distribution of soybean meal and artificial N fertilizer can also be considered as being environmental hot spots. The contribution from ‘Slaughterhouse’ (which includes transport from farm to slaughterhouse) is only 4%, while the transport from the slaughterhouse to Harwich Harbour only contributes 1%. The contribution from lorry (distance: 126 km) is seven times as high as from ship (distance: 619 km).

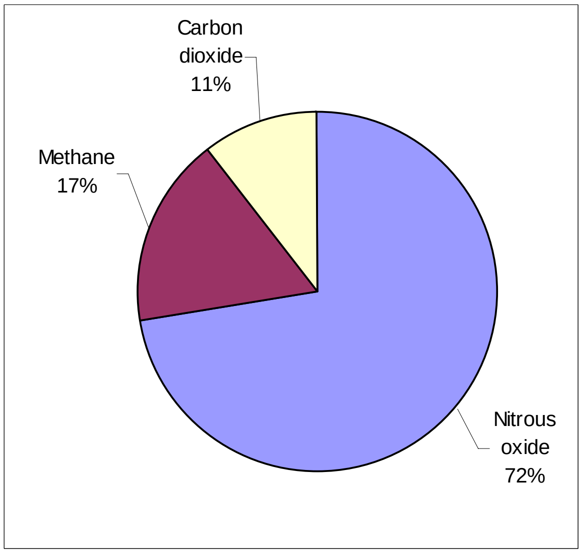

Fig 3. Types of greenhouse gases emitted from the fattening pig farm. Unit: CO_2 eq

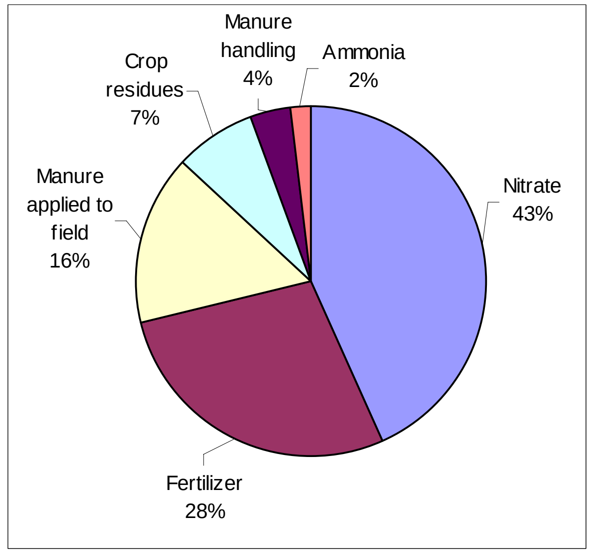

Fig. 4. Sources of nitrous oxide emissions on the fattening pig farm.

Eutrophication potential: The contribution to eutrophication is 71% from the fattening pig farm and 22% from the ‘weaners’. Out of this. 69%, 28% and 3% comes from nitrate, ammonia and phosphate, respectively. Nitrate and phosphate are leached from the fields. Ammonia is primarily emitted from the animal house, during storage in slurry tanks and under and after application of the slurry to fields.

Acidification potential: The fattening pig farm and the production of weaners contribute 60% and 26%, respectively, of the emitted acidifying substances. Ammonia from the farms amounts to 83% of the acidifying substances emitted from the product chain of Danish pork. The ammonia emitted from the fattening pig farm comes from slurry in the stable (38%), storage of slurry in slurry tanks (12%), application of slurry (20%) and N fertilizer (11%) to the fields and from the crops (19%). ‘Feed’ accounts for 6% of the acidifying substances emitted, and fertilizer and ‘energy’ purchased by the fattening pig farm account for 3% and 1%, respectively. The contribution from ‘slaughterhouse’ and ‘transport to UK’ are 3% and 1%, respectively, primarily related to energy use.

Photochemical smog potential: Substances contributing to photochemical smog primarily come from refining and combustion of fossil fuel. ‘Feed’ is the largest contributor (29%), with soybean meal being more important than feed grains. ‘Fattening pig farm’ contributes 22% of the photochemical smog potential, and out of this 21% can be ascribed to methane emission and the rest to volatile organic compounds emitted from diesel combustion.

4.2 Land use

The land used for production of 1 kg Danish pork equals 9.1 m^2year. Of these, 3.4 m^2year are used for soybean cultivation in Argentina. So an increase in demand of 1 kg Danish fattening pig implies the use of 3.4 m2 in Argentina and 5.7 in Denmark during one year.

5 Discussion

Focus on the environmental impact of food transport has increased (e.g. Pretty et al. 2005; Saunders et al. 2006), but according to our results the food transport is of minor importance for all impact categories. Considering global warming potential, only 1% of the greenhouse gases emitted can be related to the transport from the slaughterhouse to Harwich Harbour. The transport by lorry (126 km) emitted seven times as much greenhouse gas as the transport by freighter oceanic (619 km). Thus, to use ‘food miles’ as an environmental indicator would be misleading, if the type of transport was not specified. Much of the Danish pork is also sold on the Japanese market, and a sensitivity test showed that if the pork was sailed to Tokyo Harbour in Japan (distance: 21,693 km) instead of Harwich Harbour, the global warming potential per kg pork would increase by 5% (from 3.77 to 3.96 kg CO_2 eq.). So even though the pork is transported a long distance, the most polluting part of the product chain is still the farm and more environmental improvement could be achieved by increasing nutrient use efficiency in the livestock production than by focusing on food miles. Special attention should be given to the nitrogen-containing compounds nitrate, nitrous oxide and ammonia as these are the main contributors to eutrophication, global warming and acidification, respectively. A more efficient use of nitrogen on the pig farms is an effective means of reducing the environmental impact of pork production. There are several ways of increasing nitrogen efficiency.

For example, a manure management system where the manure is handled as slurry/liquid (as opposed to solid manure) will decrease the nitrous oxide emissions (Oenema et al. 2006), and ammonia evaporation can be reduced by covering the slurry tanks, which is already embedded in the Danish legislation. Higher N efficiency in the field can – for example - be obtained by timely application of slurry/manure and by injection of slurry into the soil. This might reduce the nitrate leaching and thus also the nitrous oxide formed during denitrification of leached nitrate. Reduced nitrate leaching from the fields also improves the nitrogen availability for the crops and thus reducing the requirement for artificial fertilizer, which accounts for 13% of the greenhouse gases emitted (table 4) from the product chain of pork. More efficient nitrogen (and phosphorus) use on the farms will, in addition to a reduction in on-farm emissions, also facilitate the reduced consumption of farm inputs, which again will improve the environmental profile of pork.

We compared our results with two relevant European studies from France (Basset-Mens & van der Werf 2005) and Sweden (Cederberg & Flysjö 2004), see table 5. Basset-Mens & van der Werf (2005) evaluated three pig production systems using the LCA methodology, and their normalised results show that the pig production systems contributed more to eutrophication, acidification and land use than to the other impact categories. In our study global warming, eutrophication and acidification were the most important, but we did not normalize the results on land use. The functional unit in the studies of Basset-Mens & van der Werf (2005) and Cederberg & Flysjö (2004) were '1 kg of pig (live weight) from farm gate' and '1 kg of bone and fat free meat' respectively. These functional units were transformed to the functional unit of this study (1 kg pork (carcass weight))', by assuming 1 kg of pig (live weight) equalled 0.763 kg pork (carcass weight), which again equalled 0.443 kg bone and fat free meat. The comparative results of Cederberg & Flysjö (2004) and Basset-Mens & van der Werf (2005) and our study are presented in table 5. Meat processing and transport from the slaughterhouse are only included in our study, but as these sections of the product chain only contribute 4-5% of the emissions to the considered impact categories (table 4), they will not change the overall conclusion of the comparison. Our results on global warming and eutrophication were 16% and 10% higher than the results of Basset-Mens & van der Werf (2005), which again were higher than the results of Cederberg & Flysjö (2004). In our study nitrous oxides emitted from the farms were the highest contributor to global warming potential, and out of this 43% (fig. 4) derived from denitrification of leached nitrate. This source of nitrous oxide emissions was not included in the calculations of Cederberg & Flysjö (2004) and is probably the reason for the discrepancy in results on global warming. In our study nitrate leached from the two farms was responsible for most of the eutrophication, with 69 and 73 kg N $ha^{-1}year^{-1}$ leached from the fattening pig and weaner pig farms respectively. In comparison, only 22-26 kg N $ha^{-1}year^{-1}$ was leached as nitrate in the study of Cederberg & Flysjö (2004), and this might be the reason for the lower level of eutrophication. In both the Swedish and French studies the pig houses and feed production were identified as environmental hotspots, which is in good accord with our results.

Table 5. Comparison of characterized results. Functional Unit: One kg pig from farm gate (carcass weight)

	Global warming	**Eutrophication**	**Acidification**	**Land use**
Unit	**kg CO_2-eq**	**g NO_3-eq**	**g SO_2**	**(m^2 year)**
Cederberg & Flysjö (2004)	2.6	170	37	7.2
Basset Mens & van der Werf (2005)	3.0	274	57	7.1
This study	3.5	301	56	9.1

Some authors have argued that consequential LCA modelling requires more data (Curran et al. 2005, Heijungs & Guinée submitted). However, it is the experience from this study, that several situations would have required more data-gathering and would have been much more time-consuming if attributional modelling had been used. An example is feed: Several types of feed ingredients are imported to Danish pig farms. Special feed mixtures are produced for different life stages of the pigs, because sows with piglets have different vitamin, nutrient and energy requirements compared with – for example - fattening pigs. Each feed mixture contains several ingredients. A simple screening of fodder imported to a pig farm producing both fattening pigs and weaners showed over 20 different feed ingredients (e.g. barley, maize) . In this study, we have focused only on the marginal concentrate, soybean meal (Dalgaard et al. submitted; Schmidt & Weidema submitted), which reduced data needs considerably, even though palm oil production forms parts of the LCA-data for soybean meal. From our point of view it seems logical to use the consequential LCA modelling to answer the question: ‘What happens if the demand for pork increases?' The question is highly relevant because at a global level pork production is increasing, and so is its pressure on the environment (Steinfeld et al. 2006). An increased production of pork implies an increased production of feed, electricity, fertilizer, etc. and the main challenge in consequential LCA modelling is to determine the marginal suppliers of these inputs consistently. As we have shown in this article, the identification of marginal suppliers can be improved and facilitated by already existing economic models such as ESMERALDA.

6 Conclusion and recommendations

It proved possible to obtain consequential LCA data for an agricultural product by using representative farm data and to identify affected farms using the econometric sector model ESMERALDA.

The environmental impact from an increased demand of one kg Danish pork (carcass weight) delivered to Harwich Harbour is 3.77 kg CO_2-eq. for global warming potential, 319 gram NO_3-eq. for eutrophication potential, 59 gram SO_2-eq. for acidification potential and 1.27 gram ethene-eq. for photochemical smog potential. The land use is 9.1 m^2year per kg Danish pork delivered to Harwich Harbour.

Main contributors to all impact categories were the pork farm and the weaner farm. The largest contributor to global warming was nitrous oxide from farm fields . The contribution from transport from farm via slaughterhouse to Harwich Harbour was only 1% of the greenhouse gases emitted. Considering the farm inputs (e.g. feed, fertilizer), the highest contribution came from feed. The largest contributor to eutrophication was nitrate emitted from the fields on the pig farms, and largest contributor to acidification was ammonia evaporated during slurry handling and storage. The nitrogen-containing compounds nitrate, nitrous oxide and ammonia were responsible for a large part of the environmental impact. An increase in the nitrogen use efficiency on the pig farms will obviously improve the environmental profile of Danish pork.

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4

Environmental assessment of Digestibility Improvement Factors Applied in Animal Production: Use of Ronozyme® WX CT Xylanase in Danish Pig Production

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LCA Case Studies

Environmental Assessment of Digestibility Improvement Factors Applied in Animal Production: Use of Ronozyme® WX CT Xylanase in Danish Pig Production

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Abstract

Background, Aims and Scope. Many feed ingredients are not fully digested by livestock. However, the addition of digestibility-improving enzymes to the feed can improve the absorption of e.g. energy and protein and thereby enhance the nutrient value of the feed. Feed production is a major source of environmental impacts in animal production, and it is obvious to assume that enzyme supplementation can help to reduce the environmental impact of animal production. The purpose of the study is, therefore, to assess and compare the environmental burdens of the supplements and compare them with the savings made when enzymes are used in animal production. The properties of enzymes vary considerably and the study takes as its starting point a particular enzyme product, Ronozyme WX CT. Ronozyme WX CT is a xylanase which depolymerises xylans (a group of dietary fibres found in cereal cell walls) into smaller units. The product is a widely accepted means of improving the energy value and the protein digestibility of pig and poultry feed. The study relates to Ronozyme WX CT used for fattening pigs produced in Denmark.

Methods. Lifecycle assessment is used as the analytical method, and Ronozyme WX CT production and reductions in feed consumption are modelled using SimaPro 7.0.2. Data on Ronozyme WX CT production are derived from Novozymes' production facilities in Denmark. Other data are derived from the literature and from public databases. Changes in feed consumption are determined by modelling in AgroSoft® feed optimisation software. Guidelines from the Intergovernmental Panel on Climate Change (IPCC) are used to estimate reductions in the emission of greenhouse gases resulting from reduced manure generation and changed manure composition.

Results. The study shows that the use of Ronozyme WX CT to increase the nutritional value of pig feed is justified by major advantages in terms of reduced potential contribution to global warming, acidification and photochemical ozone formation and reduced use of energy, and in most cases also nutrient enrichment and use of agricultural land. Ronozyme WX CT (xylanase) is often used together with Ronozyme P5000 CT (phytase) and together the two products can contribute considerably to reducing a broad range of environmental impacts from pig production.

Discussion. Reduced contribution to acidification and nutrient enrichment is partly driven by reduced feed consumption and partly reduced N-emissions with manure resulting from reduced protein content of the feed. Sensitivity analyses of a range of parameters show that the observed advantages are generally robust although exact magnitudes of environmental advantages are associated with much variation and uncertainty. It should, however, be noted that changes (e.g. of feed prices) may turn contributions to nutrient enrichment and use of agricultural land into trade-offs.

Conclusions. Improvement of energy and protein value of pig-feed by application of Ronozyme xylanase and following feed savings reduces impact on environment per unit of pig-meat produced, and the enzyme product contributes to a sustainable development the Danish pork meat supply.

Recommendations. Digestibility-improving enzymes are a promising means of reducing the environmental impact of pig production. The greenhouse gas reducing potential of Ronozyme WX CT in Danish pig production has been estimated at 5% and in the order of 4 million tons of CO_2 equivalents if the results are extended to the whole of Europe. Use of Ronozyme WX CT is driven by overall cost savings in animal production, and it is therefore recommended that digestibility-improving enzymes are given more attention as a cost-efficient means of reducing greenhouse gas emissions.

Keywords: Animal production; biotechnology; Danish pig production; digestibility; environment; enzymatic; enzyme; feed; manure; pig; Ronozyme; xylanase

Introduction

Many feed ingredients are not fully digested by livestock. However, the addition of digestibility-improving enzymes to the feed can improve the absorption of the feed components and enhance the value of the feed as a source of energy, protein and other nutrients (Schäfer et al. 2007, Ullmann's 2003). Depending on enzyme and animal type, the immediate advantages to the farmer are reduced feed expenditure and improved animal health. Previous environmental studies have, however, shown that feed production is a major source of environmental impacts in for instance pig production (Eriksson et al. 2005), and it is obvious to assume that, in addition to the immediate advantages to the farmer, there may also be a range of environmental advantages, as less nutrient supplementation is needed and less feed is consumed per unit of animal produced. Nielsen and Wenzel (2007) have previously documented the environmental benefits of substituting a nutrient (inorganic phosphorus) in feed with an enzyme (phytase) and it could be interesting to expand the scope to other types of digestibility-improving enzymes and assess to what extent the observations apply also to the feed-saving enzymes.

The purpose of the present study was, therefore, to assess and compare the environmental burdens of the supplements that are associated with the use of a feed-saving enzyme with the savings obtained due to the better digestion.

Digestibility improving enzymes vary considerably (Schäfer et al. 2007) and the study takes as its starting point a particular enzyme, namely the enzyme product Ronozyme WX CT. Ronozyme WX CT is a an industrially produced endo-1,4-beta-xylanase which depolymerises xylans (a group of non-starch polysaccharides) into smaller units. The enzyme product has a positive effect on the feed conversion efficiency in non-ruminants (DSM 2005, Tybirk 2005) because these animals do not have the endogenous enzymes needed to degrade dietary fibre constituents such as xylans. Pigs and poultry are non-ruminants and the use of xylanase is widely accepted as a means of improving the energy and protein value of their diet. The enzyme product, which is derived from *Thermomyces lanuginosus spp*. and produced by submerged fermentation by Novozymes A/S (Denmark), and marketed by DSM Nutritional Products (Switzerland) is authorised in the EU (Commission Regulations (EC) No 1332/2004 and No 2036/2005). The focus of the current investigation is on the environmental implications of using Ronozyme WX CT in pig feed.

1 Method

Animal feed for commercial animal production must meet a range of requirements in terms of nutrient value to the animals at the lowest possible price, and is often optimised in terms of composition by computer modelling by feed producers. The study addresses changes in environmental impact when a feed producer switches from a commercial feed product without Ronozyme WX CT to a commercial feed product with Ronozyme WX CT. The composition of commercial feed products with and without Ronozyme WX CT is modelled using AgroSoft® WinOpti, a software product used in practice in animal feed optimisation. Reductions in CH_4 emissions as a consequence of reduced manure generation coming from reduced feed consumption are determined by modelling (IPCC 2006) based on the following assumptions. Dry matter content of feed: 86% (Christiansen 2005), average dry matter digestibility coefficient of the feed: 83% (Poulsen et al. 2001), maximum methane-producing capacity = 0.45 m^3 CH_4 kg^{-1} of VS excreted (Western Europe), methane conversion factor: 17% (liquid manure) and 2% (solid manure), percentage of Danish pig manure treated as liquid and solid are 92% and 8% respectively (Poulsen et al. 2001), methane density: 0.67 kg/m^3. The environmental assessment is based on principles described by Wenzel et al. (1997). Modelling has been facilitated using the LCA software SimaPro 7.0.2. A marginal and market-oriented approach is taken in the study, and co-product issues are handled by system expansion (see Wenzel 1998, Weidema et al. 1999, Ekvall and Weidema 2004). Critical assumptions and uncertain data are addressed by sensitivity analysis.

2 Scope

2.1 Functional unit (fu)

The function of Ronozyme WX CT is to break down cereal fibres and make more constituents of the feed available to the animals, so that feed composition can be adjusted and feed consumption can be reduced without compromising meat production. The functional unit of the study is, therefore, a certain (but unspecified) quantity of meat and the study provides an assessment of the changes in environmental impact and resource consumption when one switches from producing one ton of feed without Ronozyme WX CT to a nutritionally equivalent but reduced quantity of feed with an altered ingredient composition with Ronozyme WX CT added. The amount of meat produced is not quantified because it is unnecessary for the assessment and depends on a range of variable factors such as animal breed, production conditions, etc. (**Fig. 1**). The functional unit is, however, in the order of 280 kg meat (carcass weight), because about 270 kg feed is used to produce one pig of around 100 kg and about 76% of the pig is meat (Christiansen 2005).

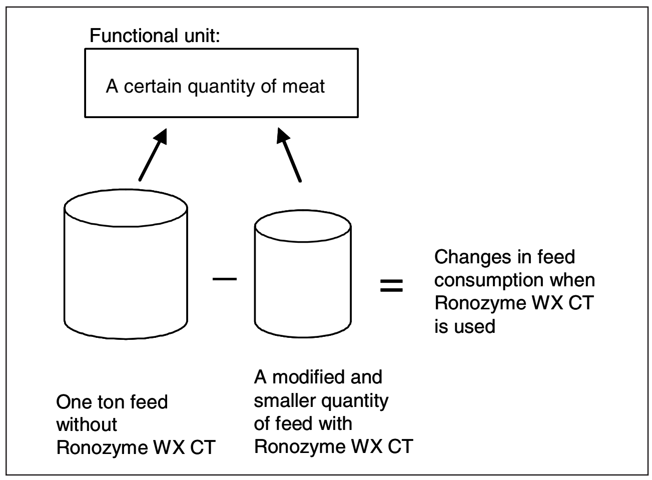

Fig. 1: Illustration of the relationship between the functional unit of the study and the changes in feed composition under consideration

2.2 Animal category, geographical scope and time perspective

The study addresses fattening pigs (25–100 kg) produced in modern production systems in Denmark. Ronozyme WX CT can be added to or removed from the feed without capital investment, and introduction of the product in the feed has no long-term implications.

2.3 Indicators

The main environmental issues in the system under consideration are deemed to be potential contributions to global warming, acidification, nutrient enrichment and photochemical ozone formation and the essential resource consumptions are considered to be energy use and use of agricultural land, and indicators are included accordingly. Characterisation is based on Eco-indicator 95 v. 2.03. Normalisation refers to Denmark (Stranddorf et al. 2005).

2.4 System boundaries

The main system boundaries of the study are shown in **Fig. 2.** The figure illustrates that the use of Ronozyme WX CT increases barley consumption and reduces soy, wheat and animal fat consumption. The changes in feed composition and the reduced feed consumption lead to reduced manure generation per unit of meat produced and reduced N content of the manure. N in manure serves as a fertiliser in crop production, and reduced N content increases the need for alternative N-fertiliser to maintain crop production. Artificial fertiliser production is regulated by fertiliser demand (in contrast to manure, which is determined by animal product demand) and artificial N-fertiliser meets the need. Emissions of N_2O and NH_3 from the stable and from manure storage systems decrease when the N content of the manure decreases. Similarly, emissions of N_2O, NH_3 and NO_3 from the agricultural land receiving the manure decrease because N emissions from manure are greater than emissions from artificial N-fertiliser. Carbon contained in the manure is to some extent converted into CH_4 during storage in the stable and in manure storage systems, and reduced manure generation per produced unit of meat results in reduced CH_4 emissions. The details are given in the following sections.

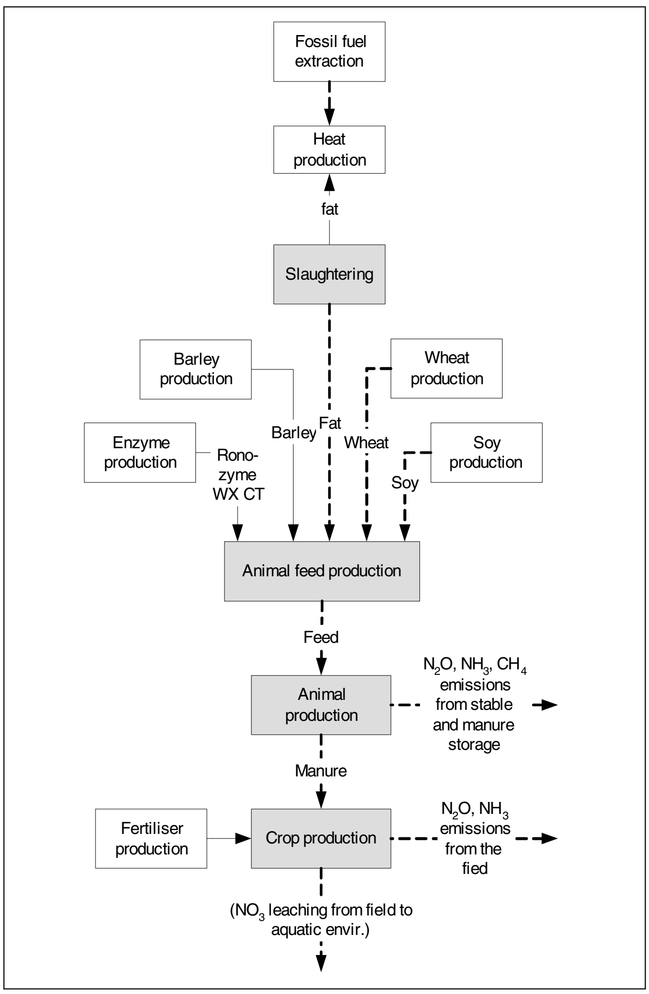

Fig. 2: Main system boundaries of the study of Ronozyme WX CT used in pig production. Full arrows indicate increased material streams, while dotted arrows indicate reduced material streams. Processes indicated with white boxes are influenced by Ronozyme WX CT application and included in the study. Grey boxes indicate processes which are independent and hence not included in the study. Reductions in emissions of N_2O, NH_3 and CH_4 to the atmosphere from stable/manure storage and field due to changed and reduced feed consumption are included in the study. Reductions in the leaching of NO_3 (in brackets) are not

All environmentally significant processes are included in the study, except reductions in NO_3 emissions to the aquatic environment due to the reduced N content of the manure. The reason is that the available data basis and modelling tools for estimating the reduction in NO_3 emissions from farmland are considered too uncertain to be included in the base case of the study. The significance of neglecting NO_3 emissions in the study has, however, been addressed in sensitivity analyses, based on the best available models.

3 Data basis and modelling

3.1 Production of Ronozyme WX CT

Ronozyme WX CT is a granulated enzyme product produced in Novozymes' factories in Denmark. The assessment of the product includes all heat, electricity and water consumptions in production and waste management and 99% (w/w) of ingredients. Modelling is based on 2005 recipes. Modelling follows Nielsen et al. (2007) except that the marginal source of electricity has been switched from natural gas to coal (Behnke 2006).

3.2 Modification of feed and reduction of feed consumption

Pig feed can be composed in many different ways and the implications of Ronozyme WX CT application can be numerous. The study addresses economically optimised animal feed in a reference situation with no Ronozyme WX CT application and in an altered situation where Ronozyme WX CT is applied, based on feed prices in 2006 (**Table 1**).

Changes induced by the adjustment of feed composition in Table 1 are determined by subtracting the quantities of individual ingredients used to produce one ton of feed in the scenario with Ronozyme WX CT application from similar ingredients in the reference situation without Ronozyme WX CT application. Increased energy and protein values of the feed lead to a reduction in soy meal and animal fat requirements on the one hand and an increase in the use of barley on the other.

Changes induced by reduced feed consumption: The National Committee for Pig Production (Tybirk 2005) proposes a 3% feed saving when commercial xylanase products are added to pig feed. The effects of xylanase application vary with diet, animal gender, growth stage of animals, etc., and it has been assumed, conservatively, in the study that the use of Ronozyme WX CT reduces the feed demand by only 2.5%. The changes induced by reduced feed consumption influence all ingredients equally and feed savings are determined accordingly.

Table 1: Pig feed composition with and without Ronozyme WX CT application and changes induced by adjustment of feed composition and 2.5% reduced feed consumption

Feed ingredients	Without Ronozyme WX CT	With Ronozyme WX CT	Changes induced by adjustment of feed composition	Changes induced by 2.5% reduced feed consumption	Total change	Inc. in model
	kg·ton feed^{-1}	kg·ton feed^{-1}	kg·fu^{-1}	kg·fu^{-1}	kg·fu^{-1}	
Barley	324	375	51	–9.4	42	Yes
Wheat	350	350	0	–8.8	–8.8	Yes
Wheat bran	50	50	0	–1.3	–1.3	No
Peas	30	30	0	-0.75	–0.75	Yes
Soy bean meal	182	151	–31	–3.8	–35	Yes
Animal fat	26	5	–21	–0.13	–22	Yes
Molasses	10	10	0	–0.25	–0.25	No
Ronozyme WX CT	0	0.20	0.20	–0.0050	0.20	Yes
Others	28	29	~ 0	–	-	No
Total	1,000	1,000	0	25	-	-

The total changes are determined by adding the changes resulting from adjustment of feed composition and the changes resulting from reduced feed consumption.

3.3 Modelling of feed ingredients

Data on barley, wheat and peas are from LCA food (2003), and are based on representative data for the Danish agricultural sector (Dalgaard et al. 2006). Data on barley refer to spring barley. Data on soybean meal refer to soybean production in Argentina (LCA food 2003). It is assumed that palm oil is the marginal type of vegetable oil (Schmidt and Weidema 2007) and that a marginal reduction of soy oil production resulting from a reduced soy bean meal demand leads to increased palm oil production. Data on palm oil production are derived from Ecoinvent (2005).

Animal fat from slaughterhouses is in excess in the Danish market and marginal fat is used in energy production (Hvid et al. 2005). A marginal reduction in animal fat consumption for pig production resulting from the use of Ronozyme WX CT is, therefore, likely to displace other energy sources (see Fig. 2). It is assumed that animal fat displaces an equivalent quantity of fuel oil (w/w), that the heat value of fat and fuel oil are similar and that emissions are the same. The only difference taken into account is thus that CO_2 from fuel oil is fossil and contributes to global warming, whereas CO_2 from animal fat does not because it comes from a non-fossil source.

Wheat bran is a relatively cheap feed ingredient and it is economically attractive to increase the use thereof when Ronozyme WX CT is applied because the xylanase increases the digestible protein and energy value of the product. Marginal wheat bran is, however, already used in animal feed production (Cerealia 2005) and since production is inherently determined by wheat grain production it is judged that production will remain unchanged independently of an increased demand induced by Ronozyme WX CT application. Wheat bran use has therefore been fixed at the reference level during modelling of feed composition. The result is an increase in barley consumption instead. Similar market constraints are assumed for molasses production (a co-product from sugar production) and similar modelling restrictions are applied for this ingredient.

The group of 'others' refers to a variety of vitamins and minerals which are virtually independent of Ronozyme WX CT application and hence disregarded in the assessment.

3.4 Estimation of changes in emissions from the stable and the manure storage

Evaporative emissions of N_2O, NH_3 and CH_4 are regarded as the essential emissions from stable and manure storage in Denmark where washout to the aquatic environment to a large extent is controlled. N_2O and NH_3 emissions are functions of N excretions with pig manure which is in turn a function of feed composition and feed consumption per unit of meat produced. Reductions in N excretions as a consequence of Ronozyme WX CT application is to a large extent driven by changes in feed consumption and a to lesser extent feed savings and has been estimated at 1.98 kg N per functional unit as specified in **Table 2.** The change in the N content of manure is proportional to the change in the pro-

Table 2: Estimation of changes in the protein content of feed, N content of manure and emissions of N_2O and NH_3 from stable and manure storage systems. The protein content of the feed is determined by modelling in AgroSoft. All data are provided per functional unit

Protein in feed (kg)			Change of N cont. of manure [b] (kg)	Change of emissions to air (g)	
Without Ronozyme WX CT	With Ronozyme WX CT	Change		N_2O-N	NH_3-N
165.9	153.5 [a]	–12.4	–1.98	–4.6	–400

[a] inc. 2.5% feed saving
[b] N content in protein ~ 16% (Sawyer et al. 1994)

Table 3: Estimation of changes in manure production and CH_4 emissions from stable and manure storage systems. All data are provided per functional unit

Total feed consumption (kg)			Change in manure production	Change in CH_4 emission
Without Ronozyme WX CT	With Ronozyme WX CT	Change	(kg dry matter)	(g)
1,000	975	–25	–3.7	–180

Table 4: Estimation of changes in direct N_2O and NH_3 emissions from the field, when manure is replaced by artificial fertiliser

Emission and source	Fraction of N emitted	Change $g{\cdot}fu^{-1}$	Total change $g{\cdot}fu^{-1}$
N_2O-N from manure	0.01	–16	–4
N_2O-N from artificial fertiliser		+12	
NH_3-N from manure	0.02	–32	–8
NH_3-N from artificial fertiliser		+24	

tein content of the feed because meat production and hence N retention in the pig is fixed (see Fig. 1). N undergoes a range of processes in the stable and manure storage systems. The fraction of excreted N which is emitted as N_2O from stable and manure storage systems is estimated at 0.23% (Dalgaard et al. 2006, IPCC 2006) and the percentage which is emitted as NH_3 is estimated at 20% (Andersen et al. 1999).

CH_4 emission is a function of manure production which is in turn a function of feed consumption per unit of pig meat produced. It is assumed that a 2.5% feed saving (see Table 1) leads to a 2.5% reduction in manure generation, and changes in methane emission per functional unit are determined by modelling (IPCC 2006) as specified in Section 1. The results are shown in **Table 3**. Enteric methane formation in pigs is low compared with methane formation from manure (IPCC 2006) and changes in enteric methane formation are ignored in the study.

3.5 Estimation of compensatory artificial fertiliser application

Manure is used as fertiliser in crop production and reduced input of N with manure to agricultural land as a result of Ronozyme WX CT application leads to an increased demand for artificial fertiliser to maintain crop yields. It is assumed that 75% of N in manure is utilised by the crops (Danish Plant Directorate 2006) and compensatory artificial fertiliser has been estimated at 75% of the remaining N content of the manure after evaporation in stable and manure storage (see Table 2), i.e. a change of 1.2 kg $N{\cdot}fu^{-1}$. Calcium ammonium nitrate (Patyk and Reinhardt 1997) is used to represent the additional artificial fertiliser.

3.6 Estimation of changes in N_2O and NH_3 emissions from the field

The substitution of manure with artificial fertiliser leads to changes in direct N_2O and NH_3 emissions from the field. The total change in N lost by evaporation from stable and manure storage is around 400 $g{\cdot}fu^{-1}$ (see Table 2) and the remaining N content in manure has been estimated at 1.6 kg fu^{-1}. The estimation of changes in N_2O and NH_3 emissions is based on emission factors from IPCC (2006) and Andersen et al. (1999) respectively. Results are shown in **Table 4**.

3.7 Other effects of Ronozyme WX CT application

The use of xylanase has a range of positive effects on the health and growth of animals (Schäfer et al. 2007). These effects are ignored in the study because they are difficult to quantify in the present context. Better health and faster growth of animals reduces the impact of pig production per unit of meat produced, and it is considered likely that this simplification leads to a slight underestimation of the environmental advantages of Ronozyme WX CT application.

3.8 Transport

All feed ingredients except soybean meal are assumed to be produced locally and transportation is ignored because it is considered negligible. Soybean meal is assumed to be produced in South America and a rough estimate of transportation is included in the assessment (ocean freighter; 10,000 km). Ronozyme WX CT is transported from factory to feed mill via a network of supply stations and a conservative estimate of transportation is included in the assessment (lorry; 1,400 km).

3.9 Data quality assessment

Modelling of **enzyme** production is based on very detailed production information. Modelling of upstream processes is based partly on data from specific suppliers and partly on generic sources. The most important data are considered to be up to date and representative and the quality of the assessment of enzyme products is considered good.

Modelling of **feed ingredients** is to a large extent based on detailed studies of agricultural production in Denmark and the data refer to marginal suppliers in Denmark. Data are reasonably up to date and quality is generally considered good although some emissions, particularly to air and water, are associated with much variation and uncertainty.

4 Results

Changes in environmental impacts and resource consumptions when Ronozyme WX CT is used in pig feed and the composition of the feed is changed and feed consumption is reduced are shown in **Fig. 3**.

Fig. 3 shows that the impacts induced by Ronozyme WX CT production are low compared with the reduction in impacts obtained by feed savings and change of the feed composition and that considerable environmental improvements can be achieved in terms of all considered impact categories when Ronozyme WX CT is used to increase the energy and protein value of the feed.

The reduced contribution to global warming is to a large extent driven by the reduced use of soybean meal (reduced N_2O emissions from soy fields), but reductions in wheat, fat and emissions of N_2O and CH_4 from stable, manure storage and field also play a role. The increased use of barley (see Table 1) and to a lesser extent the use of fertiliser to compensate for missing N in animal manure counteract the reduced contribution to global warming, but the reduction in impact due to savings of fat, wheat and soybean meal is greater than the increase in impact due to greater barley and fertiliser consumption.

Reduced fossil energy consumption is primarily driven by the reduced use of fat in the feed (see Fig. 1) and the reduced contribution to photochemical ozone formation is primarily driven by a combination of the reductions in soy and animal fat use.

The reduced contribution to acidification and nutrient enrichment is largely driven by reduced emissions of NH_3 from the stable and manure storage (see Table 2). Contributions to acidification from increased barley and fertiliser use are small compared with the fall in contributions due to reduced NH_3 emissions. Contribution to nutrient enrichment from barley is considerable, and adjustment of the feed leads to a slight increase in impact in terms of nutrient enrichment. However, the increase is too small to outweigh the savings from saved feed, and the net result is a slight reduction in contribution to nutrient enrichment.

Reduced use of agricultural land is primarily (85%) driven by reductions in feed consumption, whereas adjustment of the feed drives the largest reductions in contributions to global warming (80%), acidification (90%), photochemical smog formation (65%), and fossil energy use (95%).

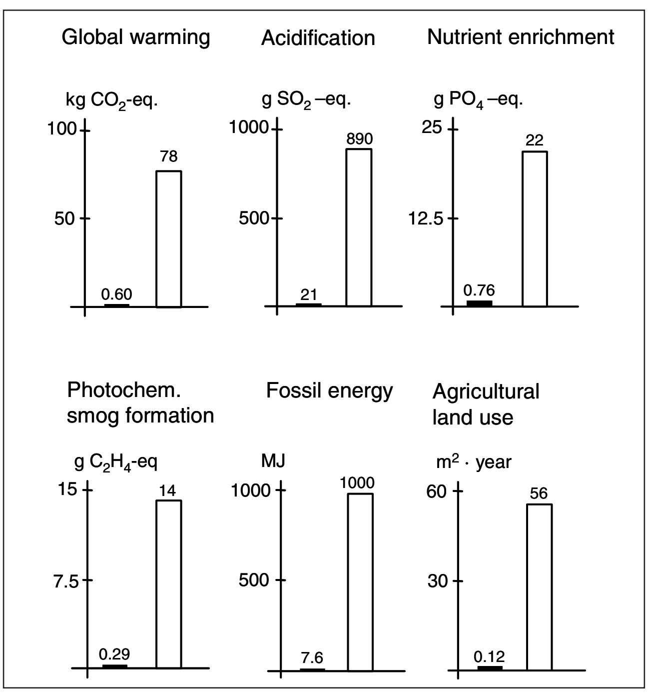

Fig. 3: Increased environmental impact potentials resulting from Ronozyme WX CT production (black bars) compared with reduced environmental impact potentials resulting from changed and reduced feed consumption (white bars). All changes are given per functional unit. Fossil energy refers to primary energy resources (lower heat value)

5 Sensitivity Analyses

A number of assumptions and simplifications have been made during the study. Some of most the important are subject to sensitivity analyses in the following in order to assess the robustness of the results.

Increased feed yield: It has been estimated conservatively that the feed saving obtained by the use of Ronozyme WX CT is 2.5%. The National Committee for Pig Production proposes a 3% feed saving (Tybirk 2005), but it is possible that the saving could be even higher. The full assessment has therefore been made with feed savings of 3.0, 3.5 and 4.5%. The results show that the estimated environmental benefits of enzyme application increase linearly when the feed saving increases. Feed saving has a considerable effect on the avoided contributions to nutrient enrichment and agricultural land use, whereas the effect on other impact categories is more limited.

Estimates of emissions from agricultural land used in crop production are uncertain and play an important role in the study. A sensitivity assessment in which emissions of the most important components (N_2O, NH_3, N_2O, NO_3, and PO_4,) from the most important fields (soy, barley and wheat fields) are varied individually and together has therefore been carried out. Uncertainty is estimated at ±20% for NH_3 and NO_3 and ±35% for PO_4 and N_2O based on Kristensen (2004) and Halberg et al. (2007). The outcome shows that the variation in NH_3, NO_3, PO_4 and N_2O emissions from the agricultural land lead to linear changes in environmental impacts and demonstrates that environmental Ronozyme WX CT is clearly advantageous in terms of all considered impact categories except nutrient enrichment independently of variations in field emissions. In most cases, Ronozyme WX CT application is also advantageous in terms of nutrient enrichment but a few cases demonstrate that it is possible that nutrient enrichment could be a trade-off if emissions from the barley field have been underestimated.

It has been assumed that palm oil is the marginal source of vegetable oil. It can, however, not be ruled out that other oils, most likely rapeseed oil, will become the marginal vegetable oil in the future and the assessment has been carried out with rapeseed oil as the marginal vegetable oil. Rapeseed production contributes more to the considered impacts than palm oil and the impact reduction obtained by Ronozyme WX CT application is therefore also smaller.

Nutrient enrichment turns out to be a trade-off of Ronozyme WX CT application whereas it remains a clear advantage for all other impact categories.

Ronozyme WX CT application increases barley consumption (see Table 1) and the environmental assessment referred to spring barley (Section 3.3). Barley is, however, also available as winter barley and the assessment has also been carried out with winter barley, since there is no indication that the one type of barley is more appropriate than the other. Except with regard to nutrient enrichment, winter barley contributes less to the environmental impacts and resource consumption under consideration than spring barley, and the switch from spring barley to winter barley emphasises the advantages of Ronozyme WX CT application slightly in terms of all impact categories except nutrient enrichment. Ronozyme WX CT application turns into a slight disadvantage in terms of nutrient enrichment.

Determination of feed composition with and without Ronozyme WX CT application is based on current feed prices. Feed prices are, however, subject to fluctuations and a series of assessments have therefore been carried out with realistic (+/–20%) variation of the relative prices of the three most important feed ingredients (barley, wheat and soybean meal). The results show that feed ingredient prices have a considerable influence on the feed composition with and without Ronozyme WX CT and hence the impact of Ronozyme WX CT application. Ronozyme WX CT application remains, however, a clear advantage in terms of all considered impact categories except nutrient enrichment and agricultural land use in a few scenarios. The reason is that the advantages of feed savings in some cases are exceeded by disadvantages from switching to crops with larger contributions to nutrient enrichment and smaller yield per area of agricultural land used. Agricultural land is currently used to produce energy crops with a CO_2 avoidance efficiency in the order of 2 to 4 tons of $CO_2 \cdot ha^{-1}$ (WTW 2006). The worst case of increased agricultural land use as a result of Ronozyme WX CT application has a greenhouse gas avoidance efficiency of 8 tons of CO_2-eq$\cdot ha^{-1}$. The agricultural land use induced by Ronozyme WX CT application is, thus considered environmentally efficient in any case. Reduced contribution to global warming varies between 30 and 80 kg CO_2 eq.$\cdot fu^{-1}$ and the average of all observations (incl. the base case, see Fig. 3) is 53 kg CO_2 eq.$\cdot fu^{-1}$.

Emissions from stables, manure storage systems and fields are dependent on a range of factors in pig production and estimates are subject to much variation and uncertainty. Sensitivity analyses where the reduction in emissions has gradually been reduced to zero show that contributions to global warming and photochemical smog formation are rather insensitive to variations (Ronozyme WX CT application is a clear advantage in terms of contribution to global warming and photochemical smog formation independently of avoided emissions), whereas contributions to acidification and particularly nutrient enrichment are very sensitive. It is estimated that NH_3 emissions (the main source of avoided contributions to acidification and nutrient enrichment) can be reduced by 50% with optimised practice (Sommer et al. 2006) and a test with 10% NH_3 emissions (instead of 20%, see Section 3.4) has been performed. The results show that Ronozyme WX CT application remains a clear advantage in terms of acidification whereas nutrient enrichment turns into a small trade-off.

Reductions in NO_3 emissions from the field have not been included in the assessment (see Fig. 1) because the data and the modelling basis were considered too poor. NO_3 contributes to nutrient enrichment and ignoring NO_3 emissions in the assessment may have lead to an underestimation of the advantages of the use of Ronozyme WX CT in terms of this impact category. Based on Dalgaard et al. (2006) rough estimates of the reduction in NO_3 (in both sandy and sandy loam soil) has been established and the assessment including the reduction in NO_3 emissions has been performed. Although the results are uncertain, they indicate that the reduction in contributions to nutrient enrichment as a result of Ronozyme WX CT application is underestimated considerably in the base case as well as in the above-mentioned sensitivity assessments, and that Ronozyme WX CT application is advantageous also in terms of nutrient enrichment in all cases except where the marginal vegetable oil switches from palm oil to for instance rapeseed oil. If the marginal vegetable oil switches to rapeseed oil, the normalised trade-off of nutrient enrichment appear to be at the same level as normalised advantages in terms of e.g. global warming.

6 Discussion

The present study has addressed Ronozyme WX CT application for pigs produced in Denmark, and the results are directly comparable with results on Ronozyme P5000 CT (phytase) application (Nielsen and Wenzel, 2007). The two enzyme products are often used together and their environmental advantages at realistic usage levels (150 g Ronozyme P5000 CT ton feed^{-1} and 200 g Ronozyme WX CT ton feed^{-1}) have been compared. The results show that the two product supplement each other in terms of environmental improvement: the xylanase's main environmental improvement potential rests in reducing contributions to global warming, acidification and photochemical smog formation and the phytase's main potential rests in reducing contributions to nutrient enrichment. Acknowledging that xylanase reduces N emissions and phytase reduces P emissions, any nutrient-enrichment-trade-offs induced by xylanase application are exceeded by much larger nutrient-enrichment-reductions obtained by phytase application when the two products are used together.

7 Conclusions

The present study shows that application of Ronozyme WX CT xylanase as a means of increasing the nutritional value of pig feed is justified by major benefits in terms of reduced contributions to global warming, acidification and photochemical ozone formation and reduced use of energy and, in most cases, also nutrient enrichment and use of agricultural land. Sensitivity analyses indicate that Ronozyme WX CT application in certain cases may lead to increased nutrient enrichment or increased use of agricultural land because the enzyme induces a switch to crops with higher impacts in terms of the two impact categories. Normalised trade-offs in terms of nutrient enrichment are the same order of magnitude as reductions in contributions to global warming and application of Ronozyme WX CT is justified unless the weighting given to nutrient enrichment is higher than or equal to global warming. Ronozyme WX CT (xylanase) is often used together with Ronozyme P5000 CT (phytase). The phytase product has major potential for nutrient enrichment reduction and any trade-offs in terms of nutrient enrichment from xylanase application are by far exceeded by the savings obtained by phytase application. Considerable reductions of greenhouse gas emissions obtained by Ronozyme WX CT application justifies the additional agricultural land use observed in a few cases.

8 Perspectives

The use of Ronozyme WX CT saves on average 52 kg CO_2 equivalents per functional unit (53–0.60 kg CO_2 eq.·fu^{-1}, Section 5 and Fig. 3), i.e. around 185 g CO_2-eq.·kg $meat^{-1}$ (carcass weight ex farm), see Section 2.1. The total greenhouse gas emissions from pig production are in the order of 3.5 kg CO_2-eq.·kg $meat^{-1}$ (carcass weight ex farm; Dalgaard et al. 2007), and the study indicates that the use of Ronozyme WX CT has the potential to reduce the contribution to greenhouse effect from Danish pig production by around 5% (up to 8% and down to 3% depending on feed prices).

Sensitivity analyses have shown that avoided contribution to global warming has a limited sensitivity to parameter variation (except feed prices), and the total potential for reduction of greenhouse gas emissions from European pig production has been roughly estimated at 4 million tons CO_2-eq. if Ronozyme WX CT were implemented in all feed (assuming (i) that the effect of xylanase use on other pig groups (sows, weaners, etc.) is the same as for fattening pigs, (ii) that 270 kg feed is consumed per pig (Christiansen (2005), (iii) that 286 million pigs are produced annually in Europe (FAOSTAT 2007), and (iv) that 52 kg CO_2 eq. is avoided per ton of feed on average (see Fig. 1 and above). Xylanases (of any brand) have currently penetrated about 30% of the feed market in Europe, and a considerable environmental improvement potential is within reach. The use of Ronozyme WX CT is driven by overall cost savings in animal production, and it is therefore recommended that digestibility-improving enzymes should be given more attention as cost-efficient means of reducing greenhouse gas emissions.

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5

Slurry technologies and their potential for environmental improvements

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Slurry technologies and their potential for environmental improvements

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(Manuscript in preparation)

Introduction

Denmark produces 25 million pigs yearly (Danish Meat Association, 2007), and has one of the world's largest pig productions per unit (Steinfeld et al., 2006). In 2006 Danish farmers produced 598 pigs per km^2 and 4.7 pigs per person (Statistics Denmark, 2007). Danish pig farms have undergone large structural changes over the last decades (Kristensen & Hermansen, 2002), as also seen in many other countries (OECD, 2003). Pig farms are becoming larger but also more concentrated in specific geographical areas (Kristensen & Hermansen, 2002).

The centralisation of pig farms in specific regions of Denmark affects the environment, with slurry being one of the most important factors. Slurry contains nutrients and may therefore be considered a valuable fertiliser source for growing crops. Slurry can also be used in the production of biogas which substitutes fossil energy and thereby reduces the emission of greenhouse gases. On the other hand, some of the nitrogen in slurry volatilises as, for example, ammonia or nitrous oxide, which affects the environment negatively. Ammonia has an acidifying effect and nitrous oxide is a strong greenhouse gas. Phosphate or nitrate can also be leached after application of the slurry to the crops, and thus pollute surface or ground water.

As part of Danish compliance with the EU Nitrate Directive, the use of slurry-N is limited, hence livestock farms with a high livestock density are obliged to export slurry to cash crop farms or livestock farms with a low livestock density. Furthermore, each farm has a fertiliser quota, so if more slurry-N is imported by a cash crop farm it must import commensurately less artificial fertiliser-N. Pig farmers with more than 1.4 livestock units per hectare (equals 49 fattening pigs (30-102 kg) produced per ha per year (Anonymous, 2006)), have to export slurry-N from the farm.

The Danish legislation on the use of slurry is based on the N content in the slurry, and therefore P is often applied to the fields in excess (Petersen, 2007). The P/N ratio in pig slurry is higher than in cattle slurry (Poulsen et al., 2001), and thus overfertilization with P is a larger problem on pig farms. Slurry separation produces a fibrous fraction with a P/N ratio that is higher than in unseparated slurry. Thus, if nutrients are exported, the fibrous fraction would export more P from the farm than the un-separated slurry .The fibrous fraction also contains less water, so reducing transport costs.

Many of the slurry separation plants in Denmark were established during 2006, and at the beginning of 2007 approximately 3% of the Danish slurry was separated (Birkmose, 2007). Most slurry separation plants are situated on pig farms, and an important reason why farmers invest in a slurry separation plant is that it facilitates the export of slurry from the farm (Birkmose, 2007). Slurry separation is often considered a ‘green technology’ (e.g., Anonymous (2007)), because it reduces the amount of slurry transported and decreases the P load on the pig farmer’s fields.

Slurry (or fractions from the slurry separation) can also be anaerobically digested, resulting in biogas production, which can be used for production of electricity and heat. Anaerobic digestion of slurry can take place in farm-scale plants situated on the livestock farm or in a central plant to which several farmers deliver their slurry.

The overall aim of this article is to clarify to what extent and under which circumstances slurry separation and anaerobic digestion reduce the environmental impact of pig production. More specifically the aim is to answer the following questions:

- To what extent does slurry separation reduce the amount of slurry transported out of the pig farm?
- To what extent does slurry separation decrease the P load to fields?
- How do slurry separation and anaerobic digestion of the slurry affect the level of greenhouse gases emitted?

Methodology

In order to analyse the slurry technologies’ potential for environmental improvements, data were collected at two large private pig farms and these data were used to establish a farm model to create different scenarios. The two farms used in the study were located in Jutland in Denmark, and they produced, respectively, 280 and 410 t pig slurry weekly, making them some of the largest pig farms in Denmark. During the winter 2004/2005 data on pig herd, crop area, slurry production and sale were collected from the two farms, and samples of raw and separated slurry were collected and analysed for content of P, total N, ammonia N and dry matter. Based on these data the separation efficiencies were calculated (Petersen & Sørensen, submitted). The separation efficiency is a measure of the 'mass of elements in the fibrous fraction as percentages of the unseparated slurry' (Petersen & Sørensen, submitted). E.g., if the N separation efficiency is 20-25% it means that 20-25% of the N in the unseparated slurry will end up in the fibrous fraction. A standardized farm model of a large scale pig production was established in order to assess the overall environmental benefits of different separation efficiencies and compare systems with different slurry technologies.

Establishment of farm model and scenarios

With the aim of calculating to what extent slurry separation and anaerobic digestion reduce transport, P-load on the pig farms or greenhouse gas emissions, a farm model was developed. The farm model represented a Danish pig farm producing approximately 1000 year sows, where pigs are kept on the farm until they obtain a weight of 100 kg. This corresponded to a yearly excretion of 122 t slurry-N (Poulsen et al., 2001), and based on Andersen et al. (2001) it was assumed that 22 t ammonia-N was emitted from the pighouse (17 t N) and storage (5 t N). These N flows and the P flows are presented in figure 1. The P content in the slurry was from the measurement at the two private pig farms (Petersen & Sørensen, submitted), and the measured P/N ratios in the excreted slurry were high compared the nutrient norms of pig slurry published by Poulsen et al. (2001).

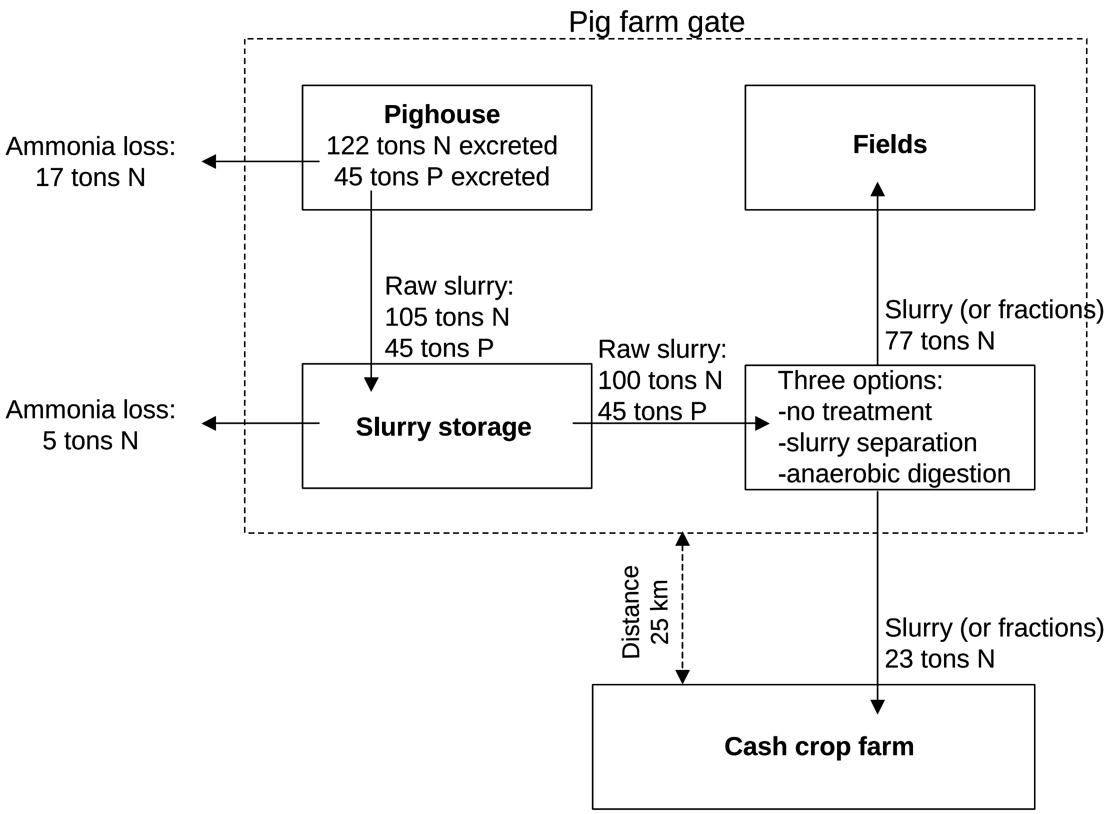

Figure 1. The N and P flows in the farm model. The N and P flows shown are the same in all scenarios. The values for the P flows to fields (at the pig farm) and to export differ amongst scenarios.

In the farm model the area was 553 ha and 23 t N was exported in order not to exceed the limit of 140 kg slurry-N applied per ha. A prerequisite for the farm model was that if the N separation efficiency was 20-25% (Møller et al., 2002) and all the fibrous fraction (with 23 t N) was exported, the area on the pig farm used for the spreading of the liquid fraction should be sufficiently large not to exceed the limit of 140 kg N per ha. Therefore the area at the pig farm in the farm model was set to 553 hectares.

The farm model represented a pig farm which was considerable larger than an average Danish pig farm (Dalgaard et al. 2006; Kristensen & Hermansen, 2002), because mainly large farms can afford the considerable economic investment that a slurry separation plant entails.

Five different scenarios were established with the farm model differing in respect to types of slurry treatment (no treatment, slurry separation, anaerobic digestion). Table 1 presents the five scenarios and the characteristics of each scenario will be described in the following.

In scenario 1 the slurry was not separated or anaerobically digested, and only raw slurry was exported out of the farm.

Table 1: Characteristics of the five scenarios

	Scenario 1	Scenario 2	Scenario 3	Scenario 4	Scenario 5
Slurry separation	No	Yes	Yes	Yes	No
Anaerobic digestion	No	No	No	No	Yes
Separation efficiency[1)]		N: 20-25%[2)]	N: 8%[3)]	N: 7%[3)]	
	-	P: 80-83%[2)]	P: 38%[3)]	P: 26%[3)]	-

1) Calculated as mass of elements in the fibrous fraction as percentages of the unseparated digested slurry.
2) Møller et al. (2002)
3) Petersen & Sørensen (submitted)

In scenarios 2, 3 and 4 the slurry was separated in a decanter centrifuge, which separated the solids from the liquid. The solid fraction (from now on designated the ‘fibrous fraction’) contained straw and fibre from the pig dung and pig hairs, while the liquid fraction contained most of the water and pig urine. The liquid fraction had high N and low P contents compared to the fibrous fraction (Møller et al., 2002; Petersen & Sørensen, submitted). The separation efficiencies in scenario 2 are from Møller et al. (2002) and these were higher than the separation efficiencies in scenarios 3 and 4 that were measured at the two private farms. The methodology used for measurements at the two private farms was explained by Petersen & Sørensen (submitted).

The two private farms and the five scenarios (presented in table 1) produced by the farm model had several characteristics in common: the farms were large, they produced both sows, piglets and fattening pigs, they had a high livestock density and were therefore obliged to export slurry-N out of the farm. At the two private farms the slurry was also anaerobically digested and the biogas was used for heat and electricity production. The heat was used in pighouses and the electricity was sold to the grid. This means that less (or no) fossil energy was used in the pighouse and the electricity replaced fossil-based electricity on the market.

Consequently, the anaerobic digestion of slurry resulted in avoided emissions of fossil CO_2. In scenario 5 the slurry was anaerobically digested, but not separated. Due to higher transparency of the results, none of the five scenarios included the combination of anaerobic digestions and slurry separation. The amounts of P applied to the fields at the pig farm and exported out of the pig farm were calculated using the nutrient contents (N and P) of the slurry, the fibrous and the liquid fraction measured at the private farms (Petersen & Sørensen, submitted).

Estimation of greenhouse gas balances in the scenarios

The greenhouse gas emissions were expected to vary between the five scenarios. In a scenario where the slurry was separated, less slurry would be exported out of the farm and therefore less fossil CO_2 emitted. Moreover a better distribution of P could be expected, and therefore less artificial P fertiliser would be used resulting in a saving of greenhouse gases emitted from the production and distribution of artificial P fertiliser. Furthermore, the energy produced from the biogas after anaerobic digestion would substitute both electricity and heat, thus resulting in avoided emissions of fossil CO_2. On the other hand, more electricity would be used in order to run the slurry separation plant, and this would result in extra greenhouse gas emissions. For all scenarios greenhouse gas balances were established based on the following assumptions: Transport of slurry (or slurry fractions) included the greenhouse gas emissions related to the transport of slurry (or slurry fractions) from the pig farm to the receiving farm. The distance was assumed to be 25 km, and the Life cycle assessment (LCA) data were taken from Ecoinvent Centre (2004, lorry size: 32 t). In accordance with Hinge (2005) it was assumed that 4.5 kWh electricity was used per tonne slurry separated. LCA data on electricity were taken from the LCAfood database (www.LCAfood.dk), and LCA data for artificial fertiliser-P from Ecoinvent Centre (2004). It was assumed that one kg slurry-P substituted one kg artificial fertiliser P. In scenario 5 the avoided emissions of fossil CO_2 were calculated according to LCA data on anaerobic digestion of pig slurry in a farm-scale biogas plant, where 1 t of slurry resulted in avoided emissions of 49 t of CO_2-eq. (Nielsen, 2004).

Results and discussion

The scenarios established using the farm model differed in terms of type and amount of slurry (or fractions) exported out of the farm and P applied per hectare at the pig farm.

The results from all scenarios are presented in table 2. In scenario 1 only raw slurry was exported from the farm because the slurry was not separated or anaerobically digested. In scenario 2 only the fibrous fraction was exported, because the N separation efficiency was high and the fibrous fraction therefore contained so much N (23 t) that the remaining liquid fraction did not exceed the application limit of 140 kg N per ha . In scenarios 3 and 4 the N separation efficiencies were lower (from the private farms) giving a lower N content in the fibrous fraction, resulting in some of the liquid fraction also having to be exported in order not to exceed the 140 kg slurry-N applied per hectare limit. In scenario 5 only anaerobically digested slurry was exported. Table 2 also shows how the amount of slurry exported was reduced when the slurry was separated. In scenario 1, 4427 t of slurry was exported yearly, while this was reduced by 37% (to 2808 t) if the slurry was separated according to the separation efficiencies in scenario 2. However, because of the lower separation efficiencies (measured at the private farms) in scenarios 3 and 4, the amount of slurry exported was only reduced by 18% (to 3636 t) and 10% (to 4002 t) in scenarios 3 and 4, respectively. Thus, even though the slurry was separated, the ‘saved’ transport of slurry in scenarios 3 and 4 was limited because the separation efficiencies were much lower compared with scenario 2. In scenario 5 the amount of slurry exported was not reduced.

Table 2: Results on slurry export and amount of P applied per ha per year in the five scenarios.

	Scenario 1	Scenario 2	Scenario 3	Scenario 4	Scenario 5
Type of slurry exported	Raw slurry	Fibrous fraction	Fibrous and liquid fraction	Fibrous and liquid fraction	Anaerobic digested slurry
Slurry exported out of pig farm, tons/year	4427	2808	3636	4002	4427
N exported out of pig farm, tons/year	23	23	23	23	23
P exported out of pig farm, tons/year	10	38	19	14	10
P applied to field at pig farm, kg P/ha	62	12	46	55	62

Table 2 also shows that 10 t P was exported in scenario 1 and this was almost quadrupled in scenario 2, because the fibrous fraction in scenario 2 contained considerably more P per unit N compared with the raw slurry in scenario 1. Slurry-P applied to fields was reduced by 82% (from 62 to 12 kg P/ha) when the slurry was separated. But in scenarios 3 and 4, where the separation efficiencies were lower, the slurry-P applied was only reduced by 11% and 26%, compared with scenario 1. However, in scenario 2 only 12 kg P per hectare was applied to fields and this is below the P removed from the soil if for example 7000 kg wheat per hectare was cultivated. So seen in a longer-term perspective, the application of 12 kg P per hectare would lead to a P deficit. However, only few pig farms suffer from P deficit, so reducing the amount of P applied to fields at the pig farms is important in order to reduce phosphate leaching. More phosphate is generally leached from Danish pig farms than from dairy farms (Dalgaard et al. 2006) and this is a threat to the aquatic environment.

There is no doubt, the weight of exported pig slurry can be considerably reduced (37% in scenario 2) if the slurry separation plant separates efficiently, and so can the slurry-P applied per hectare on the pig farm (82% in scenario 2). But with the efficiencies obtained at the two private farms the environmental advantages of slurry separation will be limited.

Another important observation is that slurry separation and anaerobic digestion do not solve the problem of the relatively large amount of ammonia lost from housing and storage. According to figure 1, 22 t N is emitted (40 kg N/ha) and this is the same for all scenarios. So if pig farms are concentrated in specific areas, the ammonia load in that area will remain the same even with separated or anaerobically digested slurry.

Figure 2 presents the greenhouse gas balances for scenarios 1, 2, 3 and 4. Scenario 5 was omitted from the figure because it is out of scale. It should be emphasized here that the greenhouse gas emissions (e.g., nitrous oxide) from the fields where the slurry (or the fractions) were applied were not included, because it was assumed that the emissions would be the same for the unseparated and the separated slurry. The greenhouse gas emissions in Figure 2 include only those arising from the treatment and transport of slurry, and the amount of avoided artificial P fertiliser.

According to figure 2, 19 t CO_2-eq. were emitted from the transport of slurry in scenario 1, whereas this was lower in the other scenarios, due to less slurry being transported (see table 2). This shows that the transport-related emissions of fossil CO_2 were reduced when the slurry was separated. The greenhouse gas emissions related to 'Electricity' were the same for scenarios 2, 3 and 4 because the amount of slurry separated was the same (20,000 t).

'Avoided artificial P fertiliser' was a negative value in all four scenarios, because the slurry (or fractions) substituted artificial fertiliser-P. In scenario 2, the high P separation efficiency resulted in avoided emissions of 66 t CO_2-eq. This avoided emission more than counterbalanced the greenhouse gas emissions from 'transport' and 'electricity'. The avoided emissions from scenarios 3 and 4 were lower, because the P separation efficiencies in these scenarios were lower (see table 1). The 'Total' in figure 2 shows a positive balance for the total greenhouse gas emission for scenario 1 (1 t CO_2-eq.). The total greenhouse gas emissions from the other scenarios were negative, meaning that they reduced the emissions of greenhouse gases compared with scenario 1. Obviously, a longer transport distance would result in higher greenhouse gas emissions. A sensitivity analysis showed that the fibrous fraction in scenario 2 could be transported 128 km before it cancelled out the saved emissions from ‘avoided artificial P fertiliser’, and thereby resulted in a net positive greenhouse gas emission. But because of the low separation efficiencies in scenarios 3 and 4, the fractions in these scenarios could only be transported 28 and 45 km, respectively, before the saved emissions from ‘avoided artificial P fertiliser’ were cancelled out. So whether the slurry separation seen in relation to no separation (scenario 1) was environmentally worse or better was highly dependent on the separation efficiency. If it was low, the amounts to be transported grew, and less artificial fertiliser P would be substituted.

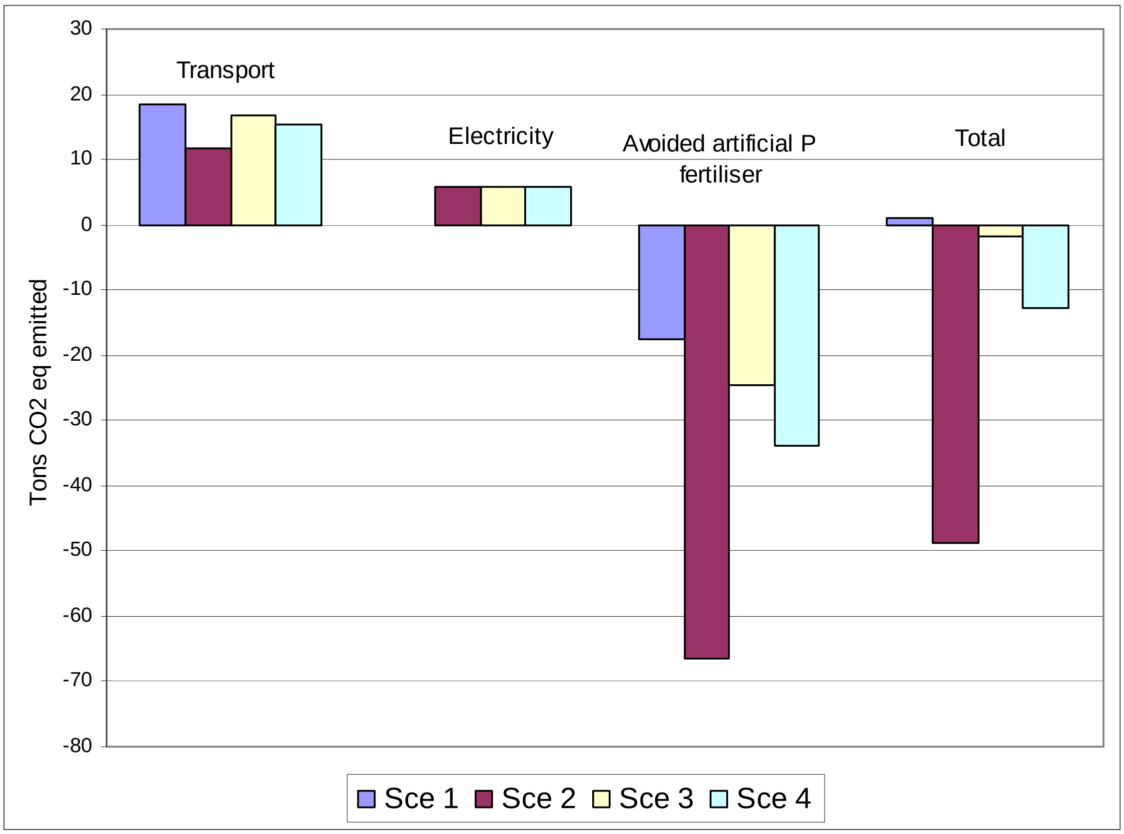

Figure 2: Greenhouse gas balances from scenarios 1, 2, 3 and 4.

In scenario 5, where the slurry was anaerobically digested, more than 1150 t fossil CO_2 was avoided due to the production of biogas-based electricity and heat (not shown). These saved emissions from the substitution of electricity and heat (‘Avoided fossil energy’) were very dominating compared to emissions from ‘transport’ (19 t CO_2-eq. (as for scenario 1 shown in figure 2)) and saved emissions from ‘avoided artificial fertiliser P’ (-18 t CO_2-eq. (as for scenario 1 shown in figure 2)). This showed that the anaerobic digestion of slurry is a very efficient technology compared with slurry separation if the goal is to reduce the greenhouse gas emissions. However, it is important that the methane losses from the biogas plant are minimized, because methane is a strong greenhouse gas, and losses of methane will result in less energy production but eventually also partly outweigh the benefit form using biogas. It should be emphasized that anaerobic digestion does not provide a better distribution of N and P between the pig and the cash crop farms, as is the case for the slurry separation scenarios presented in table 2.

Besides slurry handling the product chain of the pig also includes, for example, activities inhouse and in the field, the production and transport of artificial fertiliser and feed, heat, electricity, etc. (Dalgaard et al. submitted). Using the separation efficiencies in scenario 2 and the results from figure 2 the reduced greenhouse gas emissions arising from slurry separation were compared to the rest of the product chain of the pig. For this comparison it was assumed: i) the pig farm in the scenarios produced 2795 t live weight pigs per year and ii) the emission per kg pig (carcass weight) was 3.5 kg CO_2-eq. (Dalgaard et al., submitted). The comparison showed that if the slurry was separated, the fibrous fraction was transported 25 km, and artificial P fertiliser was substituted, the greenhouse gas emissions per kg pig could only be reduced by 0.6%. If the slurry was not separated but anaerobically digested and the biogas was used for energy production (as described above), the greenhouse gas emissions per kg pork could be reduced by 16%. So seen in relation to global warming, anaerobic digestion was not only better than slurry separation, but it also offered the opportunity to reduce the global warming potential per pig considerably. On the other hand, anaerobic digestions did not change the nutrient contents (and not the P/N ratio) in the slurry, so it did not have the same potential for reducing the P loads on the pig farm that slurry separation had.

Conclusion

Slurry separation can potentially reduce the amount of slurry transported from pig farms to cash crop farms. The P load on the pig farms can also be reduced. A greenhouse gas balance showed that the greenhouse gas emissions can be reduced if artificial P fertiliser was substituted on the cash crops farms that receive the fibrous fraction. However, if the separation efficiencies were as low as those measured on the two private farms in this study, the environmental advantages of using slurry separation will be limited. If the slurry was anaerobically digested and the biogas used for energy production, the greenhouse gas emission per pig could be reduced by 16%.

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