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Review

Comparing environmental impacts for livestock products: A review of life cycle assessments

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ABSTRACT

Livestock production has a major impact on the environment. Choosing a more environmentally-friendly livestock product in a diet can mitigate environmental impact. The objective of this research was to compare assessments of the environmental impact of livestock products. Twenty-five peer-reviewed studies were found that assessed the impact of production of pork, chicken, beef, milk, and eggs using life cycle analysis (LCA). Only 16 of these studies were reviewed, based on five criteria: study from an OECD (Organization for Economic Cooperation and Development) country, non-organic production, type of LCA methodology, allocation method used, and definition of system boundary. LCA results of these 16 studies were expressed in three ways: per kg product, per kg protein, and per kg of average daily intake of each product for an OECD country. The review yielded a consistent ranging of results for use of land and energy, and for climate change. No clear pattern was found, however, for eutrophication and acidification. Production of 1 kg of beef used most land and energy, and had highest global warming potential (GWP), followed by production of 1 kg of pork, chicken, eggs, and milk. Differences in environmental impact among pork, chicken, and beef can be explained mainly by 3 factors: differences in feed efficiency, differences in enteric CH_4 emission between monogastric animals and ruminants, and differences in reproduction rates. The impact of production of 1 kg of meat (pork, chicken, beef) was high compared with production of 1 kg of milk and eggs because of the relatively high water content of milk and eggs. Production of 1 kg of beef protein also had the highest impact, followed by pork protein, whereas chicken protein had the lowest impact. This result also explained why consumption of beef was responsible for the largest part of the land use and GWP in an average OECD diet. This review did not show consistent differences in environmental impact per kg protein in milk, pork, chicken and eggs. Only one study compared environmental impact of meat versus milk and eggs. Conclusions regarding impact of pork or chicken versus impact of milk or eggs require additional comparative studies and further harmonization of LCA methodology. Interpretation of current LCA results for livestock products, moreover, is hindered because results do not include environmental consequences of competition for land between humans and animals, and consequences of land-use changes. We recommend, therefore, to include these consequences in future LCAs of livestock products.

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Contents

1. Introduction

The environmental impact of livestock production has received increasing attention over the last years, because it appears to have a major impact on the environment (Steinfeld et al., 2006). The livestock sector increasingly competes for scarce resources, such as land, water, and energy, and has a severe impact on air, water and soil quality because of its emissions. According to the Food and Agriculture Organization, for example, the world's livestock sector is responsible for 18% of the global emission of greenhouse gases. This contribution of 18% was explained by emission of carbon dioxide from fossil-fuel combustion and deforestation, emission of methane from manure and enteric fermentation by ruminants, and emission of nitrous oxide from application of fertilizer during cultivation (Steinfeld et al., 2006).

In addition to changes in production practices, eating less or no livestock products, such as meat, is seen often as a possible solution to reduce the environmental impact of the livestock sector (Carlsson-Kanyama, 1998; Pimentel and Pimentel, 2003; Reijnders and Soret, 2003; Baroni et al., 2007). Indeed, a balanced plant-based diet can provide us with all the nutrients required for a healthy life (Appleby et al., 1999). Eating meat, however, is not only a reflection of nutritional needs, but it is also determined by taste, odour, and texture, as well as by geographical area, culture, ethics and wealth (Richardson et al., 1993). In member countries of the Organization for Economic Cooperation and Development (OECD), more than one-quarter of the energy content of an average diet still consists of animal products (FAOSTAT, 2009). A massive number of people in developing countries, moreover, are turning to this kind of diet (FAO, 2002).

So far, little attention has been paid to reducing environmental impact of the livestock sector by choosing the livestock product that is produced with the lowest environmental impact. Choosing a more environmentally-friendly livestock product in a diet possibly can mitigate environmental impact. To choose from among different types of meat or between protein from meat and eggs, for example, we need a consistent assessment of their environmental impact. Such an assessment requires a quantification of the emissions and resource use during the entire life cycle of that product. Life cycle assessment (LCA) is a generally accepted method to evaluate the environmental impact during the entire life cycle of a product (Guinée et al., 2002). Many studies have used LCA to assess the environmental impact of livestock products, such as pork, chicken, beef, milk, or eggs. The fact that these studies all used LCA as impact assessment tool offers an opportunity to assess which livestock products are most harmful to the environment. To our knowledge, no scientific overview of the LCA results of livestock products has been published.

The objective of this research was to compare the environmental impact for livestock products. We reviewed, therefore, 16 studies from OECD countries, using the LCA method.

2. Life cycle assessment

Life cycle assessment is a holistic method to evaluate the environmental impact during the entire life cycle of a product. Two types of environmental impact are considered during the life cycle of a product: use of resources such as land or fossil fuels, and emission of pollutants such as ammonia or methane (Guinée et al., 2002). Emission of pollutants contributes to impact categories, such as climate change, acidification and eutrophication of ecosystems, and human or terrestrial eco-toxicity.

LCA relates the environmental impact to a functional unit, which is the main function of a production system expressed in quantitative terms. Functional units in LCA studies of agricultural production, for example, are kg of fat–protein corrected milk, kg of grain produced or kg of meat produced (De Boer, 2003). Besides this functional unit, e.g. fat–protein corrected milk, a production system might yield another valuable output, such as meat or manure. Moreover, cultivation of wheat, for example, yields wheat grain and wheat straw. In these multiple-output situations, the environmental impact of the production system or process has to be allocated to various outputs. Three main allocation methods exist (ISO, 2006): economic allocation, physical allocation (e.g. mass allocation) and system expansion. In the case of economic allocation, the environmental impact of a production system or process is allocated to its multiple outputs based on their relative economic value. LCA results based on different allocation methods cannot be compared directly (Thomassen et al., 2008a).

2.1. Selection of LCA studies

We found 25 LCA studies from peer-reviewed scientific journals and scientific reports that examined the environmental impact of individual livestock products. To include only those LCA studies that are comparable, six selection criteria were defined:

- LCA studies from OECD countries
- LCA of non-organic systems
- LCA of systems that produce pork, chicken, beef, milk or eggs
- attributional LCA, i.e. evaluation of status quo situation
- economic allocation of multiple outputs
- at least cradle to farm-gate LCA.

Only LCA studies from OECD countries were selected, because livestock production systems in these countries differ substantially from systems in other countries. Animals in non-OECD countries often have multiple functions, such as draught power or capital asset, rather than only the one function of production of livestock products (Udo and Cornelissen, 1998; Moll, 2005). Multiple functions complicate an LCA because the environmental impact is allocated only to the main output of a system or process (Steinfeld et al., 2006; Schau and Fet, 2008; Phong et al., in preparation).

LCA studies of organic products were excluded from the analysis because livestock production is almost entirely non-organic in OECD countries (Steinfeld et al., 2006). So only few LCA studies on organic production of livestock products are published (Cederberg and Mattsson, 2000; Haas et al., 2001; Basset-Mens and Van der Werf, 2005; Thomassen et al., 2008b).

In OECD countries, pork, chicken, beef, milk and eggs are the most common livestock products in an average diet (FAOSTAT, 2009). A comparison of environmental impact of fish with, for example, meat is difficult as LCA initially was developed for land-based systems. Moreover, environmental impacts that are important for production of sea-food, such as over-fishing, use of antifouling, fuel emission from combustion at sea and seafloor ecosystem disturbance, are not accounted for in current LCA studies (Ellingsen and Aanondsen, 2006).

There are two ways of performing an LCA: consequential and attributional (Thomassen et al., 2008a). Consequential LCA studies aim at quantifying environmental consequences of a change in demand of a product. We included only attributional LCA studies, which aim at quantification of the environmental impact of a product in a status quo situation. The majority of LCAs on livestock products use attributional LCA (Thomassen et al., 2008a). Furthermore, we selected only LCA studies that apply economic allocation because this was the most common method of allocation.

Ideally, an LCA evaluates the environmental impact of a product over its entire life cycle, i.e. from cradle to grave. The majority of LCA studies, however, evaluated only the production stages until the farm gate, and left out succeeding stages, such as processing, retail, or household. We included studies, therefore, that evaluated at least all production stages until the farm gate. Studies that analyzed more production stages after the farm gate were included, however, if their results could be recalculated to cradle-to-farm-gate boundaries. To achieve this, system boundaries were recalculated in Hospido et al. (2003) and Zhu-XueQin and Van Ierland (2004).

Based on these criteria we included 16 LCA studies (Table 1), and excluded 9 LCA studies (Baumgartner et al., 2008; Ogino et al., 2004; Eriksson et al., 2005; Bennett et al., 2006; Dalgaard et al., 2006; Elferink and Nonhebel, 2007; Ellingsen and Aanondsen, 2006; Pelletier, 2008; Weidema et al., 2008). Some studies that were excluded complied with the selection criteria, but no data were available.

2.2. Comparison of LCA studies

Farming systems in LCA studies that assessed the same product differed substantially in their characteristics, e.g. animal productivity, feed composition and production period. Such characteristics might affect environmental impact (e.g. Basset-Mens and Van der Werf, 2005; Casey and Holden, 2006; Williams et al., 2006). In OECD countries, for example, systems for production of beef are heterogeneous, whereas systems for pork, chicken and egg are usually homogeneous because of their standardized production method worldwide. If a study performed an LCA of different production systems for one product then each system was included in this review (see Table 1), on the condition that they complied with the inclusion criteria.

To compare LCA results among selected studies, the functional unit (FU) was recalculated. In addition, LCA results were expressed in the same unit and were recalculated to a cradle to farm-gate LCA.

2.2.1. Calculation of functional unit

Comparison of livestock products demands an identical FU. Various meat products, for example, can be compared based on the environmental impact per kg of meat. Relative to meat, however, milk or eggs have a high water content. A comparison of the environmental impact of products such as meat, milk, or eggs, therefore, demands a different FU, not one based on 1 kg of product.

A FU depends on the function of the product, and the primary function of livestock products is to satisfy the human body's need for nutrition, especially protein (Schau and Fet, 2008). In addition to a FU of 1 kg of product, environmental impact of livestock products was analyzed also using a FU of 1 kg of protein. One might think of other, possibly more important reasons, however, why people consume livestock products, such as texture, tradition, or culture (Richardson et al., 1993). Instead of unraveling motivations of dietary choices, we choose a single objective indicator that covers all motivations: the actual consumption of livestock products. In this review, we assumed that the “nutritional need” for livestock products is represented by consumption in OECD countries. In addition to FUs being defined as “kg of product” and “kg of protein”, therefore, a third FU was defined: average daily intake (ADI) of each product in kg for an OECD country (Table 2).

To express LCA results of studies related to meat production, according to FUs described above, first we recalculated results of each study to a FU of *1 kg of Live Weight (LW)*. Second, we

Table 1
Studies included: characteristics and original global warming potential (GWP), acidification potential (AP), eutrophication potential (EP) and use of land and energy.

Study	System/study case	Country [a]	FU [b]	GWP	AP		EP		Land	Energy
				kg CO_2-e	kg	unit	kg	unit	m^2	MJ
Pork										
Zhu-XueQin and Van Ierland (2004)[c]	Conventional	NL	t protein	77,883	675	NH_3-e	2491	N-e	55,000	397,252
Basset-Mens and Van der Werf (2005)	Good agricultural practice	F	kg live weight	2.3	0.044	SO_2-e	0.021	PO_4^{3-}-e	5.4	16
Basset-Mens and Van der Werf (2005)	Red label	F	kg live weight	3.5	0.023	SO_2-e	0.017	PO_4^{3-}-e	6.3	18
Williams et al. (2006)	Heavier finishing	UK	t dead weight	6080	391	SO_2-e	97	PO_4^{3-}-e	6900	15,500
Williams et al. (2006)	Indoor breeding	UK	t dead weight	6420	507	SO_2-e	119	PO_4^{3-}-e	7300	16,700
Williams et al. (2006)	Outdoor breeding	UK	t dead weight	6330	362	SO_2-e	95	PO_4^{3-}-e	7500	16,700
Williams et al. (2006)	Conventional	UK	t dead weight	6360	395	SO_2-e	100	PO_4^{3-}-e	7400	16,700
Cederberg and Darelius (2002)	Single intensive pig farm	S	kg bone–fat-free meat	4.8	2.6	mol H^+-e	2.0	O_2-e	15	22
Blonk et al. (1997)	Conventional	NL	kg live weight	3.7	0.031	SO_2-e	0.018	PO_4^{3-}-e	–	16
Chicken										
Williams et al. (2006)	Conventional	UK	t dead weight	4570	173	SO_2-e	49	PO_4^{3-}-e	6400	12,000
Williams et al. (2006)	Free range	UK	t dead weight	5480	230	SO_2-e	63	PO_4^{3-}-e	7300	14,500
Katajajuuri (2008)	Conventional	F	t live weight	2079	35	SO_2-e	2.1	PO_4^{3-}-e	5500	16,000
Beef										
Williams et al. (2006)	100% suckler	UK	t dead weight	25,300	708	SO_2-e	257	PO_4^{3-}-e	38,500	40,700
Williams et al. (2006)	Lowland	UK	t dead weight	15,600	452	SO_2-e	153	PO_4^{3-}-e	22,800	26,800
Williams et al. (2006)	Hill and upland	UK	t dead weight	16,400	510	SO_2-e	169	PO_4^{3-}-e	24,100	29,700
Williams et al. (2006)	Non-organic	UK	t dead weight	15,800	469	SO_2-e	157	PO_4^{3-}-e	23,000	27,800
Casey and Holden (2006)	Both specialist beef farms and dairy breeds	I	kg live weight	11	–		–		–	–
Cederberg and Darelius (2002)	Conventional, dairy calves	S	kg meat	17	4.2	mol H^+-e	3.4	O_2-e	33	40
Milk										
Basset-Mens et al. (2009)	Average farm	NZ	kg milk	0.93	0.0081	SO_2-e	0.0029	PO_4^{3-}-e	1.2	1.5
Cederberg and Mattsson (2000)	Single specialised farm	S	t ECM	990	18	SO_2-e	58	NO^{3-}-e	1925	2800
Cederberg and Flysjö (2004)	Production > 7500 ECM/ha	S	kg ECM	0.87	0.010	SO_2-e	0.0038	PO_4^{3-}-e	1.5	2.6
Cederberg and Flysjö (2004)	Production < 7500 ECM/ha	S	kg ECM	1.0	0.011	SO_2-e	0.0042	PO_4^{3-}-e	1.9	2.7
Haas et al. (2001)	Intensive	G	t milk	1300	19	SO_2-e	7.5	PO_4^{3-}-e	–	2700
Haas et al. (2001)	Extensive	G	t milk	1000	17	SO_2-e	4.5	PO_4^{3-}-e	–	1300
Casey and Holden (2005)	Average Irish dairy unit	I	kg ECM	1.3	–		–		–	–
Hospido et al. (2003)[c]	Two typical Galician dairy farms	ES	1 l packaged milk	1.1	0.0085	SO_2-e	0.0053	PO_4^{3-}-e	–	6.2
Thomassen et al. (2008b)	Ten conventional commercial dairy farms	NL	kg FPCM	1.4	0.0095	SO_2-e	0.11	NO^{3-}-e	1.3	5.0
Thomassen et al. (2009)	119 dairy farms	NL	kg FPCM	1.4	0.011	SO_2-e	0.12	NO^{3-}-e	1.3	5.3
Williams et al. (2006)	Non-organic	UK	10,000 l milk	10,600	162	SO_2-e	63	PO_4^{3-}-e	11,900	25,200
Williams et al. (2006)	More fodder as maize	UK	10,000 l milk	9800	164	SO_2-e	61	PO_4^{3-}-e	11,800	23,600
Williams et al. (2006)	60% high yielders	UK	10,000 l milk	10,200	159	SO_2-e	60	PO_4^{3-}-e	11,400	24,200
Williams et al. (2006)	20% autumn calving	UK	10,000 l milk	10,300	159	SO_2-e	65	PO_4^{3-}-e	12,100	23,400
Eggs										
Mollenhorst et al. (2006)	Battery cage	NL	kg egg	3.9	0.032	SO_2-e	0.25	NO^{3-}-e	4.5	13.0
Mollenhorst et al. (2006)	Deep litter	NL	kg egg	4.3	0.057	SO_2-e	0.31	NO^{3-}-e	4.8	13.4
Mollenhorst et al. (2006)	Deep litter with outdoor run	NL	kg egg	4.6	0.065	SO_2-e	0.41	NO^{3-}-e	5.7	13.9
Mollenhorst et al. (2006)	Aviary with outdoor run	NL	kg egg	4.2	0.042	SO_2-e	0.35	NO^{3-}-e	5.1	13.7
Williams et al. (2006)	Non-organic	UK	20,000 eggs	5530	306	SO_2-e	77	PO_4^{3-}-e	6600	14,100
Williams et al. (2006)	100% cage	UK	20,000 eggs	5250	300	SO_2-e	75	PO_4^{3-}-e	6300	13,600
Williams et al. (2006)	100% free range	UK	20,000 eggs	6180	312	SO_2-e	80	PO_4^{3-}-e	7800	15,400

[a] NL = The Netherlands; F = France; UK = United Kingdom; G = Germany; I = Ireland, F = Finland; ES = Spain, S = Sweden, NZ = New Zealand.
[b] FU means functional unit; ECM = Energy corrected milk; FPCM = fat–protein corrected milk.
[c] System boundaries from cradle to grave.

Table 2
Calculation factors to determine a functional unit.

Product	kg edible product/kg live weight [a]	kg protein/kg edible product [b]	daily intake [c] (kg/person/day)
Pork	0.53	0.19	0.082
Chicken	0.56	0.19	0.074
Beef	0.43	0.19	0.060
Milk products (excl. butter)	–	0.03	0.545
Eggs	–	0.13	0.036

[a] Source: PVE, personal communication (2008).
[b] Sources: meat: Lawrie and Ledward (2006); milk: average of all studies; eggs: USDA (2009).
[c] Source: FAOSTAT (2009).

recalculated results to a FU of *1 kg of edible product*, i.e. meat, milk or egg, using fixed calculation factors (Table 2). Because the majority of the economic value of a fattening animal comes from production of meat, the environmental impact was fully allocated to the edible product, and not to non-edible products such as leather. Third, based on the protein content of each product, we recalculated results to a FU of *1 kg of protein* (Table 2). Fourth, we expressed results as a proportion of the average daily intake (ADI) of each product in kg for an OECD country.

To express LCA results related to milk and egg production, we took a two-step approach. First, we recalculated LCA results of each study to a FU of *1 kg of milk* or *1 kg of egg*, assuming an average egg weight of 62.5 g (Animal Sciences Group, 2006). Second, we expressed LCA results per kilogram of protein and per kilogram of ADI, based on information in Table 2.

2.2.2. Unit of LCA results

In the studies reviewed, the size of an environmental impact was calculated per impact category using equivalence factors. To assess the impact of production of a specific product on climate change, the studies we reviewed quantified emission of carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O). Emission of CO_2, CH_4, and N_2O were summed up based on their equivalence factor in terms of CO_2-equivalents (100-year time horizon): 1 for CO_2, 21 for CH_4, and 310 for N_2O. Only Williams et al. (2006) and Thomassen et al. (2009) used a more recent set of equivalence factors: 1 for CO_2, 23 for CH_4, and 296 for N_2O.

Studies reviewed used different units for the impact categories acidification and eutrophication (see Table 1). Acidification potential of each study was recalculated to SO_2 equivalents, whereas eutrophication potential was recalculated to PO_4^{3-} equivalents (Bauman and Tillman, 2004). Land use was recalculated to square meters (m^2 per year) and energy use to megajoules of primary energy (MJ).

3. Results and discussion

All studies evaluated five impact categories: land use, primary energy use, climate change, eutrophication and acidification, except for Hospido et al. (2003) and Casey and Holden (2005, 2006). This section includes an analysis of the observed variation among studies of the same livestock product and among studies of different products. Environmental impacts are expressed per kg product, i.e. meat, milk, or eggs, per kg of protein, and per average daily intake (ADI) of each product in kg for an OECD country.

3.1. Land use

Land use varied among livestock products (Fig. 1). With regard to meat products (pork, chicken, and beef), production of 1 kg of pork required 8.9–12.1 m^2 and 1 kg of chicken 8.1–9.9 m^2 of land, whereas production of 1 kg of beef required 27–49 m^2 of land. The large amount of land needed for beef production has two causes. First, compared with a pig or a broiler, a beef animal is less efficient in conversion of ingested energy and nutrients into edible meat (Schroeder and Titgemeyer, 2008). Second, compared with production of beef, production of pork and chicken shows relatively little land use from breeding stock due to the relatively large number of progeny produced per mother animal annually and to early sexual maturity.

One extreme value was observed among the results for beef. This high value of 49 m^2 (Williams et al., 2006) represented land use of beef production in a system where calves were bred by suckler cows. In such a system, land required for production and maintenance of the suckler cow is added to the total land requirement of beef production. The lower values, ranging from 27 m^2 to 31 m^2, represented land use of beef production in a system where calves were bred by dairy cows, in other words calves were a by-product from the dairy farm. In such a system, the majority of the land required for production and maintenance of the “mother cow” is allocated to milk production and not to beef production.

There was no clear difference in the amount of land required to produce 1 kg of pork and chicken (Fig. 1). Williams et al. (2006) quantified land use of both pork and chicken and concluded that the amount of land required to produce 1 kg of chicken was slightly lower (i.e., 8.1–9.2 m^2) than that of 1 kg of pork (i.e., 10.1–11 m^2). This slightly higher land use for pork than for chicken results from the fact that broiler chickens need less feed per kg edible meat than pigs. This finding was confirmed by Elferink and Nonhebel (2007), and Baumgartner et al. (2008). These LCA studies were not in our review because they (partly) used mass instead of economic allocation.

Compared with production of meat (pork, chicken, and beef), production of 1 kg of milk and 1 kg of eggs required little land (Fig. 1). Production of 1 kg of milk required only 1.1–2.0 m^2, whereas production of 1 kg of eggs required 4.5–6.2 m^2. Compared with meat, milk and eggs have a relatively high water content, which causes the lower land use per kg of product.

When we expressed land use per kg of protein (Fig. 2), milk production required 33–59 m^2, which overlapped with pork (47–64 m^2), chicken (42–52 m^2), and eggs (35–48 m^2); whereas beef production required 144–258 m^2. The comparative study of Williams et al. (2006) showed that 1 kg chicken protein required slightly less land (i.e., 42–48 m^2) than 1 kg of pork protein (53–58 m^2). Williams et al. (2006) also showed that production of 1 kg of milk protein (33–35 m^2) required less land than that of 1 kg of beef protein (153–258 m^2), 1 kg of pork protein (53–58 m^2) and 1 kg of chicken protein (42–48 m^2), and that there is no clear difference in

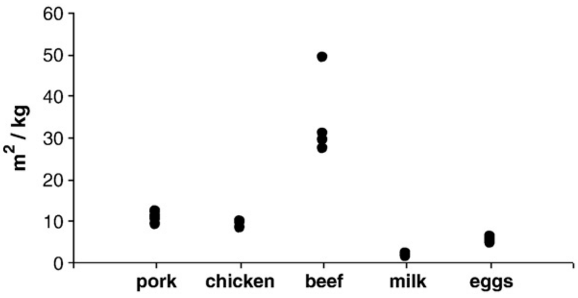

Fig. 1. Land use for livestock products (in m^2/kg of product).

land use between production of 1 kg of egg protein (41–48 m^2) and that of 1 kg of chicken protein (42–48 m^2).

When we expressed land use per amount of average daily intake of each product (ADI) (Fig. 2), the daily consumption of beef had the highest land use (1.65–2.96 m^2); followed by consumption of milk (0.62–1.1 m^2), chicken (0.60–0.73 m^2) and pork (0.73–0.99 m^2); whereas consumption of eggs (0.16–0.22 m^2) resulted in the lowest land use. Despite the high ADI of milk products (0.55 kg) compared with pork (0.08 kg), chicken (0.07 kg) or beef (0.06 kg; see Table 2), consumption of milk required less land than consumption of beef and the same amounts of land as the consumption of pork or chicken. This is because land use per kg of milk is lower than land use per kg of meat. Consumption of beef is responsible for the largest part of the land use caused by average diet in OECD countries.

The variation in land use among studies of the same product was relatively low. This low variation is because the amount of land needed to produce 1 kg of product was quantified consistently in different studies. To quantify land use, you need to estimate the amount of farm land used, i.e., on-farm land, and the amount of land required to produce all purchased inputs, such as fertilizer, pesticides, energy and feed, i.e., the so-called off-farm land. Off-farm land mainly is determined by land use for production of feed ingredients (Thomassen et al., 2008b).

The amount of on-farm land generally was estimated based on farm visits or statistical databases. The amount of off-farm land was estimated based on the amount of purchased feeds (or feed conversion in case of model calculations), the composition of these feeds, the yield per hectare of different feed ingredients, the production process of different feed ingredients and economic allocation values. The amount of purchased concentrates was estimated from farm visits or national data bases. Average composition of purchased feed generally was obtained from feed companies, whereas yield per ha generally was based on national data sets or information from FAOSTAT (2009). Differences in land use of feed ingredients were due to differences in local production circumstances, national differences in processing of feed ingredients or distinct assumptions for economic allocation of by-products in feed.

3.2. Use of fossil energy

Fig. 3 shows energy use (in MJ) for livestock products expressed per kg of product. Production of 1 kg of pork used 18–45 MJ, which overlapped with energy use of chicken (15–29 MJ) and beef (34 to 52 MJ). The estimate of 45 MJ for 1 kg of pork from Zhu-XueQin and Van Ierland (2004), however, was based on an energy analysis of Pimentel (1992), which was not obtained according to the LCA methodology (i.e., no allocation). Without this estimate for energy use of pork, energy use for pork ranged from 18 to 34 MJ, which was lower than that for beef and overlapped with that for chicken.

As with land use, based on the comparative study of Williams et al. (2006), the amount of energy used to produce 1 kg of chicken (15–18 MJ) was lower than that of 1 kg of pork (23–24). This merely results from the fact that broilers have lower feed conversion than fattening pigs. This finding was confirmed by LCA results of Baumgartner et al. (2008) based on a combination of mass and economic allocation.

The ratio of the average energy use for beef production to pork production was 1.4, whereas the ratio of land use for beef to pork was 3.1. The difference in energy use between beef and pork is smaller than the difference in land use because the relative share of concentrates in feed for beef cattle in systems studied was lower than that for fattening pigs and broilers (Vellinga et al., 2008). Production of roughage generally occurred locally, and, therefore, required relatively little energy for transport. Concentrate ingredients in pig and poultry feed, however, originated from all over the world, and, therefore, required more energy for transport.

Because of the relatively high water content of eggs and milk, energy use per kg of eggs and, especially, of milk, were lower compared with meat (Fig. 3).

When we expressed energy use in terms of protein (Fig. 4), production of milk required 37–144 MJ/kg, pork 95–236 MJ/kg, chicken 80–152 MJ/kg, and eggs 87–107 MJ/kg, whereas beef production required 177–273 MJ/kg. Williams et al. (2006) showed that production of 1 kg chicken protein required less energy (80–96 MJ) than 1 kg of pork protein (119–129 MJ). In addition, they showed that production of

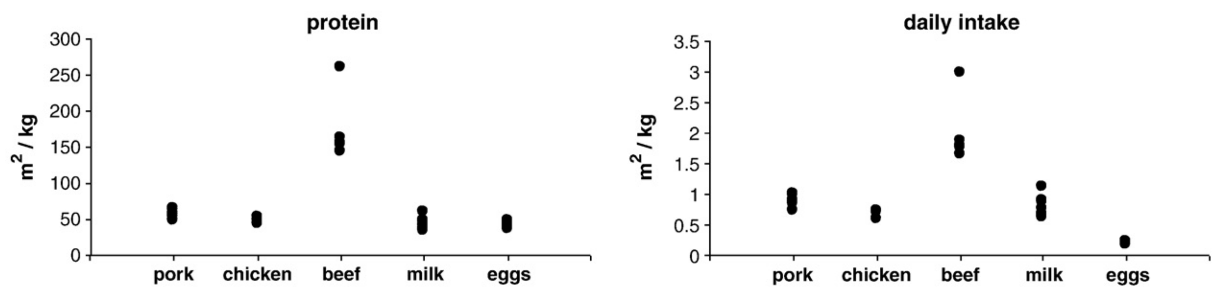

Fig. 2. Land use for livestock products, in m^2 per kg of protein or per average daily intake of each product.

Fig. 3. Energy use for livestock products, in MJ per kg of product.

Fig. 5. Global warming potential for livestock products, in CO_2-e expressed per kg of product.

1 kg of milk protein required less energy (67–68 MJ) than that of 1 kg of beef protein (187–273 MJ). Production of 1 kg of egg protein (87–95 MJ) used the same amount of energy as production of 1 kg of chicken protein (80–96 MJ).

When we expressed energy in terms of ADI (Fig. 4), daily consumption of pork and beef was responsible for the largest part of the energy use for an average diet in OECD countries.

3.3. Climate change

Fig. 5 shows global warming potential (GWP) for livestock products in CO_2-equivalents (CO_2-e) per kg of product. For livestock products from monogastric animals, generally, N_2O was responsible for the largest part of the GWP, whereas for products from ruminants N_2O and CH_4 were equally important. For all livestock products, CO_2 appeared to be the least important greenhouse gas.

Production of 1 kg of pork resulted in 3.9–10 kg CO_2-e and production of 1 kg of chicken in 3.7–6.9 kg CO_2-e, whereas production of 1 kg beef resulted in 14 to 32 kg CO_2-e. Several factors explain the differences in GWP among pork, chicken, and beef.

First, energy use for beef was highest, followed by pork and chicken (Fig. 3). Emission of CO_2 was directly related to combustion of fossil energy because CO_2 emission from changes in land use or from the carbon stock in the soil was not included.

Second, CH_4 emission per kg meat from ruminants was higher than from monogastric animals. Emission of CH_4 from monogastrics originated mainly from manure, whereas CH_4 emission from ruminants originated from manure and from enteric fermentation processes in the rumen. Enteric methane emission in ruminants explains about 75% of the CH_4 emission per animal, whereas manure management explains about 25% (Klein Goldewijk et al., 2005). The higher emission of CH_4 from ruminants compared with monogastrics also explained why for milk and beef CH_4 and N_2O are the major greenhouse gases, whereas for pork, chicken and egg N_2O is the major one.

Third, the amount of feed needed per kg of meat is higher for ruminants than for monogastric animals (Schroeder and Titgemeyer, 2008). During cultivation and transport of feed emission of greenhouse gases occur, especially CO_2 and N_2O. Therefore, emission of CO_2 and N_2O per kg product is higher for ruminants than for monogastrics.

Fourth, compared to production of beef in a suckler system, production of pork and chicken has a relatively low GWP from breeding stock due to the relatively large number of progeny per mother animal annually. When beef calves are bred by dairy cows, this argument is not relevant.

The greatest variation was observed among GWP values for production of beef (Fig. 5). A reason for this is that beef is produced in a wide range of production systems. Highest values represented systems using calves from suckler cows. Lowest values represented systems using calves from dairy cows, where GWP related to maintenance of the “mother cow” was allocated to milk production. The lowest GWP for beef (15 CO_2-e/kg) was derived from Cederberg and Darelius (2002), who studied a single farm in Sweden that fattened Holstein bull calves bred by dairy cows. These calves were raised without grazing and nearly all fodder, except for concentrates (9% of the total dry matter), was produced on-farm.

Relatively little variation was observed among GWP values for production of pork, chicken and milk. Differences in GWP assessments among LCA studies for pork or chicken resulted mainly from differences in estimates of N_2O. N_2O is formed during denitrification of nitrate in the soil. This

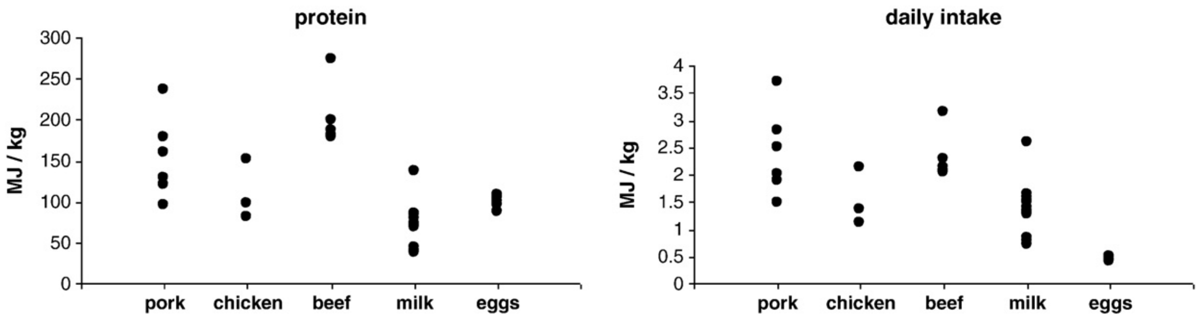

Fig. 4. Energy use for livestock products, in MJ per kg of protein or per average daily intake of each product.

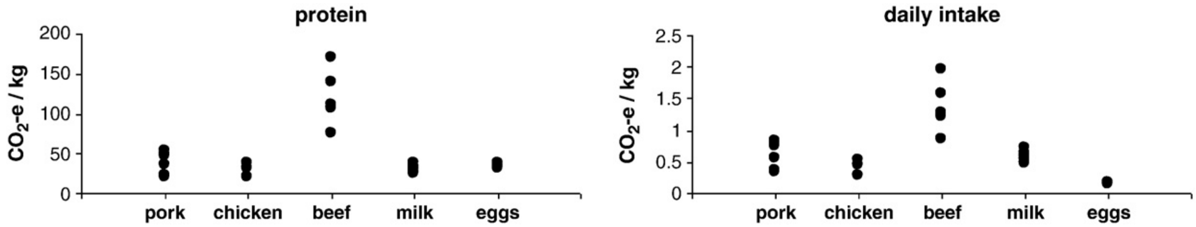

Fig. 6. Global warming potential for livestock products, in CO_2-e per kg of protein or per average daily intake of each product.

process of denitrification highly depends on soil conditions, such as soil type and groundwater level and, therefore, differs among and within countries (Schils et al., 2007). Differences in estimates of N_2O resulted from the fact that some studies used N_2O emission factors that depended on soil conditions, whereas other studies used generic N_2O emission factors from IPCC (2006).

Because of the relative high water content of milk and eggs, GWP of milk (0.84–1.3 CO_2-e) and of eggs (3.9–4.9 CO_2-e) was lower compared with meat (Fig. 5). When we expressed GWP in terms of protein (Fig. 6), production of milk had a range of 24–38 CO_2-e/kg, which overlapped production of pork (21–53 CO_2-e/kg), chicken (18–36 CO_2-e/kg) and eggs (30–38 CO_2-e/kg), whereas production of beef resulted in a GWP of 75–170 CO_2-e/kg. The comparative study of Williams et al. (2006) showed that 1 kg protein from chicken had a lower GWP (30–36 CO_2-e) than from pork (47–49 CO_2-e) or eggs (32–38 CO_2-e), and was comparable to that from milk (28–31 CO_2-e).

When we expressed GWP in terms of ADI (Fig. 6), we concluded that the major impact of the OECD consumption pattern resulted from consumption of beef; followed by pork, chicken and milk; and then by eggs.

3.4. Acidification and eutrophication

To assess the impact of production of a specific product on acidification, the studies we reviewed quantified emission of acidifying gases ammonia (NH_3), sulfur dioxide (SO_2), and nitrogen oxide (NO_x). Estimates for acidification potential (AP) showed larger variation among LCA studies of the same livestock product than estimates for land use, energy use, and GWP (Fig. 7). The AP of pork, for example, varied from 43 to 741 g SO_2/kg, with a coefficient of variation of 80%, whereas the coefficient of variation of pork for land use was 10%, for energy use 31% and for climate change 30%.

All studies showed that acidification was caused mainly by emission of NH_3, so that differences in AP can be explained by differences in NH_3. Emission of NH_3 results from manure in the housing and storage facilities, during grazing, and during application of fertilizer on the field. Emission of NH_3 emission is determined by feed ration, type of housing, manure storage facility and manure application technique, and climatic conditions, such as temperature and air velocity (Monteny et al., 2002; De Boer et al., 2002). In contrast with emission of NH_3, emission of CH_4 is affected mainly by feed ration, and not by type of housing, manure storage facility or climatic conditions (Tamminga et al., 2007). Therefore, estimates of NH_3 emission showed larger variation among studies compared with estimates of CH_4 emission.

Moreover, LCA studies used different NH_3 emission factors for different types of housing, manure storage facilities and application techniques, whereas for CH_4 emission generally IPCC reference values were used (IPCC, 2006).

Studies showed that eutrophication potential (EP) was caused mainly by emission of NH_3, and leaching or run-off of NO_3^- and PO_4^{3-}. Like AP, EP of the same product showed large variation (Fig. 7). Leaching of NO_3^- and PO_4^{3-} depends on climatic and soil conditions, and can differ largely among countries or even among regions within the same country (Schils et al., 2007). This partly explains the large variation observed in EP among similar products. Moreover, leaching and run-off of NO_3^- and PO_4^{3-} are difficult to quantify and actually unknown for many situations. Leaching/run-off of NO_3^- and PO_4^{3-} have been quantified as percentage of N or P fertilizer applied, as a fixed value per ha or based on a field nutrient balance. These differences in methodology also contributed to differences in LCA results observed.

3.5. General discussion

For land use, energy use and climate change, we can conclude that production of 1 kg of beef protein had the highest impact, followed by pork, whereas chicken had the lowest impact. This conclusion is based on results of the life cycle of

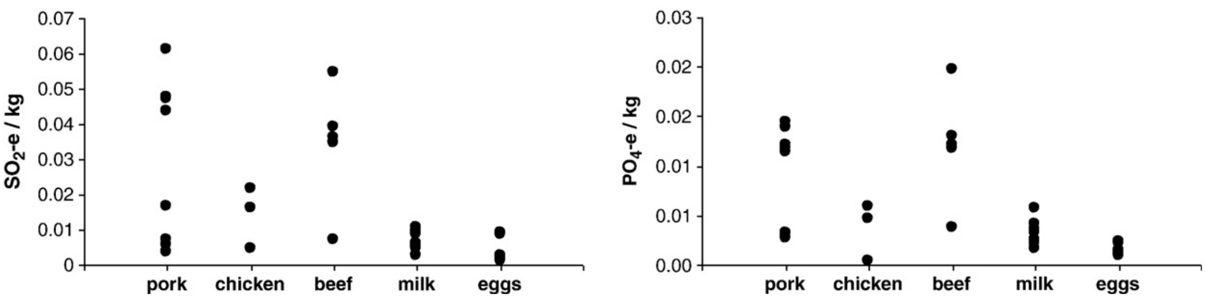

Fig. 7. Acidification potential (AP, in kg SO_2-e) and eutrophication potential (EP, in kg PO_4^{3-}-e) for livestock products, per kg of product.

meat production until the product left the farm gate. We know that during the post-farm-gate stages of production of meat, such as processing, packaging, retail and household, there is an additional environmental impact (Berlin, 2002; Hospido et al., 2003; Zhu-XueQin and Van Ierland, 2004; Katajajuuri, 2008). Differences in this additional impact among meat products might affect our conclusion. Based on literature, we do not, however, expect large differences in environmental impact of post-farm-gate stages among different meat products. Post-farm-gate stages were responsible for about 30%–40% of the total energy use and for 20%–25% of the GWP of production of pork or chicken, whereas these stages showed a minor contribution to land use, eutrophication, and acidification (Zhu-XueQin and Van Ierland, 2004; Katajajuuri, 2008).

This review did not show consistent differences in environmental impact per kg of protein in milk, chicken, pork or eggs. Conclusions regarding impact of pork or chicken versus impact of milk and eggs require further harmonization of LCA methodology and additional comparative studies (Roy et al., 2009). Only one study compared environmental impact of meat versus milk and eggs. This study showed that production of 1 kg of milk protein had lower GWP than the production of 1 kg of beef protein, and used less land and energy than production of 1 kg of beef, pork, and poultry protein. Literature showed that energy use and global warming potential in post-farm-gate stages of milk products, such as cheese, is smaller compared with meat products, i.e., 10% for climate change and 20% for energy use (Berlin, 2002; Van Middelaar, submitted for publication). The fact that the additional impact during post-farm-gate stages is smaller for milk products than for meat products enlarges the differences found between milk protein and meat protein.

Based on this review of LCA results we could propagate substitution of red meat by white meat. Compared with rations of ruminants, however, the ration of pigs and poultry contain relatively more products, such as cereals, that humans could consume directly (Vellinga et al., 2008). Direct consumption of these cereals by humans is ecologically more efficient than consumption of meat produced by animals fed with these cereals, because most of the energy is lost during conversion from plant to animal product (Goodland, 1997). Environmental consequences of this competition between humans and animals for cereals are not incorporated in current LCAs of livestock products (Garnett, 2009). So only in a situation were enough land is available worldwide to produce cereals required for all humans and livestock, we can conclude to substitute red meat by white meat.

To avoid competition between humans and animals for products such as cereals, we could stimulate to feed by-products from other agricultural activities and the human industry to monogastric animals, and to feed grass from marginal land to ruminants. A correct evaluation of the value of by-products or marginal products to feed our animals, however, should also consider possible alternative use of these products for, e.g., production of bio-energy.

Another aspect that is not included sufficiently in current LCAs of livestock products are CO_2 emissions associated with changes in land use or the carbon stock in the soil. Pigs and poultry, more so than ruminants, consume products, such as soy bean meal, whose cultivation is said to be a major driver of deforestation (Nepstad et al., 2006). Allocation of CO_2 emission associated with deforestation, however, is complicated by the fact that deforestation is a complex, dynamic process resulting, not only from cultivation of feed crops such as soy beans, but also from logging and grazing of livestock. Further research, therefore, is needed to assess how significant CO_2 emissions associated with changes in land use or the carbon stock in the soil affect GWP of livestock products.

Finally, a consumer's choice among different types of meat or protein from chicken or milk does not only depend on environmental impact of its production, but also on other sustainability issues such as animal welfare, product quality and cost price.

4. Conclusion

A large number of LCA studies of livestock products have been published. This review yielded a consistent ranging of results for use of resources land and energy, and for climate change. No clear pattern was found, however, for eutrophication and acidification. To gain insight into eutrophication and acidification potential of livestock products more comparative LCA studies are needed.

Production of 1 kg of beef used most land and energy, and had highest GWP, followed by production of 1 kg of pork, chicken, eggs, and milk. Differences in environmental impact among pork, chicken, and beef can be explained mainly by three factors: utilization of nutrients and energy in feed, differences in enteric CH_4 emission between pigs and chicken, and cattle, and differences in reproduction rates. The impact of production of 1 kg of meat (pork, chicken, beef) was high compared with production of 1 kg of milk and eggs, because of the relatively high water content of milk and eggs. Production of 1 kg of beef protein also had the highest impact, followed by pork protein, whereas chicken protein had the lowest impact. This result also explained why consumption of beef was responsible for the largest part of the land use and GWP in an average OECD diet. Only one study compared impact of meat versus impact of milk and eggs. Conclusions regarding impact of meat versus impact of milk and eggs require additional comparative studies and further harmonization of LCA methodology. Interpretation of current LCA results for livestock products, moreover, is hindered because results do not include environmental consequences of competition for land between humans and animals, and consequences of land-use changes. We recommend, therefore, to include these consequences in future LCAs of livestock products.

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References

Animal Sciences Group, 2006. Kwantitatieve Informatie Veehouderij 2006–2007. Animal Sciences Group, Lelystad, The Netherlands.

Appleby, P.N., Thorogood, M., Mann, J.I., Key, T.J., 1999. The Oxford Vegetarian Study: an overview. American Journal of Clinical Nutrition 70 (suppl. 3), 525S–531S.

Baroni, L., Cenci, L., Tettamanti, M., Berati, M., 2007. Evaluating the environmental impact of various dietary patterns combined with different food production systems. European Journal of Clinical Nutrition 61, 279–286.

Basset-Mens, C., Van der Werf, H.M.G., 2005. Scenario-based environmental assessment of farming systems: the case of pig production in France. Agriculture, Ecosystems and Environment 105, 127–144.

Basset-Mens, C., Ledgard, S., Boyes, B., 2009. Eco-efficiency of intensification scenarios for milk production in New Zealand. Ecological Economics 68, 1615–1625.

Bauman, H., Tillman, A., 2004. The Hitch Hiker's Guide to LCA. An Orientation in Life Cycle Assessment Methodology and Application. Studentlitteratur, Lund, Sweden.

Baumgartner, D.U., de Baan, L., Nemecek, T., 2008. European grain legumes — environment-friendly animal feed? Life cycle assessment of pork, chicken meat, egg and milk production. Fedearl Department of Economic Affairs DEA. Agroscope Reckenholz-Tänikon Research Station ART, Zürich, Zwitzerland.

Bennett, R.M., Phipps, R.H., Strange, A.M., 2006. The use of life cycle assessment to compare the environmental impact of production and feeding of conventional and genetically modified maize for broiler production in Argentina. Journal of Animal and Feed Sciences 15, 71–82.

Berlin, J., 2002. Environmental life cycle assessment of semi-hard cheese. International Dairy Journal 12, 939–953.

Blonk, H., Lafleur, M., Van Zeijts, H., 1997. Towards an environmental infrastructure of the Dutch food industry: exploring the information conversion of five food commodities. Amsterdam, The Netherlands IVAM Environmental Research. University of Amsterdam.

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

Casey, J.W., Holden, N.M., 2005. Analysis of greenhouse gas emissions from the average Irish milk production system. Agricultural Systems 86, 97–114.

Casey, J.W., Holden, N.M., 2006. Quantification of GHG emissions from suckler-beef production in Ireland. Agricultural Systems 90, 79–98.

Cederberg, C., Darelius, K., 2002. Using LCA methodology to assess the potential environmental impact of intensive beef and pork production (PhD Thesis). Department of Applied Environmental Science. Göteborg University, Sweden.

Cederberg, C., Flysjö, A., 2004. Life cycle inventory of 23 dairy farms in south-western Sweden. SIK Report Nr 728. The Swedisch Institute for Food and Biotechnology, Göteborg, Sweden.

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

Dalgaard, R., Halberg, N., Kristensen, I.S., Larsen, I., 2006. Modeling representative and coherent Danish farm types based on farm accountancy data for use in environmental assessments. Agriculture, Ecosystems and Environment 117, 223–237.

De Boer, I.J.M., 2003. Review. Environmental impact assessment of conventional and organic milk production. Livestock Production Science 80, 69–77.

De Boer, I.J.M., Smits, M.C.J., Mollenhorst, H., van Duinkerken, G., Monteny, G.J., 2002. Prediction of ammonia emission from dairy barns using feed characteristics. Part I: relation between feed characteristics and urinary urea concentration. Journal of Dairy Science 85, 3382–3388.

Elferink, E.V., Nonhebel, S., 2007. Variations in land requirements for meat production. Journal of Cleaner Production 15, 1778–1786.

Ellingsen, H., Aanondsen, S.A., 2006. Environmental impacts of wild caught cod and farmed salmon — a comparison with chicken. International Journal of Life Cycle Assessment 11, 60–65.

Eriksson, I.S., Elmquist, H., Stern, S., Nybrant, T., 2005. Environmental systems analysis of pig production: the impact of feed choice. International Journal of Life Cycle Assessment 10, 143–154.

FAO, 2002. World Agriculture: Towards 2015/2030. FAO, Rome, Italy.

FAOSTAT, 2009. Electronic Database of the Food and Agriculture Organization. http://faostat.fao.org/.

Garnett, T., 2009. Livestock-related greenhouse gas emissions: impacts and options for policy makers. Environmental Science and Policy 12, 491–503.

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

Guinée, J.B., Gorrée, M., Heijungs, R., Huppes, G., Kleijn, R., de Koning, A., van Oers, L., Wegener Sleeswijk, A., Suh, S., Udo de Haes, H.A., de Bruijn, H., van Duin, R., Huijbregts, M.A.J., Lindeijer, E., Roorda, A.A.H., van der Ven, B.L., Weidema, B.P. (Eds.), 2002. Handbook on Life Cycle Assessment; Operational Guide to the ISO Standards. Institute for Environmental Sciences, Leiden, The Netherlands.

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

Hospido, A., Moreira, M.T., Feijoo, G., 2003. Simplified life cycle assessment of galician milk production. International Dairy Journal 13, 783–796.

IPCC, 2006. IPCC Guidelines for National Greenhouse Gas Inventories. http://www.ipcc - nggip.iges.or.jp/public/2006gl/index.htm.

ISO, 2006. Environmental management. The ISO 14000 family of international standards ISO standard collection on CD-rom, Genève, Switzerland.

Katajajuuri, J.M., 2008. Experiences and improvement possiblities — LCA case study of broiler chicken production. Proceedings of the 6th International Conference on Life Cycle Assessment in the Agri-Food Sector. Novermber 12–14, Zurich, Switzerland.

Klein Goldewijk, K., Olivier, J.G.J., Peters, J.A.H.W., Coenen, P.W.H.G., Vreuls, H.J.J., 2005. Greenhouse gas emissions in the Netherlands 1990–2003. National Inventory Report 2005. Report 773201009/2005. Institute for Human Health and Environment, Bilthoven.

Lawrie, R.A., Ledward, D.A., 2006. Lawrie's meat science, 7th ed. Woodhead Publishing, Cambridge.

Moll, H.A.J., 2005. Costs and benefits of livestock systems and the role of market and non-market relationships. Agricultural Economics 32, 181–193.

Mollenhorst, H., Berentsen, P.B.M., De Boer, I.J.M., 2006. On-farm quantification of sustainability indicators: an application to egg production systems. British Poultry Science 47, 405–417.

Monteny, G.J., Smits, M.C.J., Van Duinkerken, G., Mollenhorst, H., De Boer, I.J.M., 2002. Prediction of ammonia emission from dairy barns using feed characteristics. Part II: relation between urinary urea concentration and ammonia emission. Journal of Dairy Science 85, 3389–3394.

Nepstad, D.C., Sticker, C.M., Almeida, O.T., 2006. Globalization of the Amazon soy and beef industries: opportunities for conservation. Conservation Biology 20, 1595–1603.

Ogino, A., Kaku, K., Osada, T., Shimada, K., 2004. Environmental impacts of the Japanese beef-fattening system with different feeding lengths as evaluated by a life-cycle assessment method. Journal of Animal Science 82, 2115–2122.

Pelletier, N., 2008. Environmental performance in the US broiler poultry sector: life cycle energy use and greenhouse gas, ozone depleting, acidifying and eutrophying emissions. Agricultural Systems 98, 67–73.

Phong, L. T., De Boer, I.J.M., Udo, H.M.J., in preparation. Life cycle assessment of food production in integrated agriculture–aquaculture systems in the Mekong Delta, Vietnam.

Pimentel, D., 1992. Energy inputs in production: agriculture. In: Fluck, R.C. (Ed.), Energy in Farm Production. Elsevier, Amsterdam.

Pimentel, D., Pimentel, M., 2003. Sustainability of meat-based and plant-based diets and the environment. The American Journal of Clinical Nutrition 78, 660S–663S suppl.

PVE, personal communication, 2008. Productschappen Vee, Vlees en Eieren (PVE). Personal communication August 25th, 2008.

Reijnders, L., Soret, S., 2003. Quantification of the environmental impact of different dietary protein choices. The American Journal of Clinical Nutrition 78, 664S–668S suppl.

Richardson, N.J., Shepherd, R., Elliman, N.A., 1993. Current attitudes and future influence on meat consumption in the U.K. Appetite 21, 41–51.

Roy, P., Nei, D., Orikasa, T., Xu, Q., Okadome, H., Nakamura, N., Shiina, T., 2009. A review of life cycle assessment (LCA) on some food products. Journal of Food Engineering 90, 1–10.

Schau, E.M., Fet, A.M., 2008. LCA studies of food products as background for environmental product declarations. International Journal of Life Cycle Assessment 13 (3), 255–264.

Schils, R.L.M., De Haan, M.H.A., Hemmer, J.G.A., Van den Pol-Van Dasselaar, A., De Boer, J.A., Evers, G.A., Holshof, G., Van Middelkoop, J.C., Zom, R.L.G., 2007. Dairy wise, a whole-farm dairy model. Journal of Dairy Science 90, 5334–5346.

Schroeder, G.F., Titgemeyer, E.C., 2008. Interaction between protein and energy supply on protein utilization in growing cattle: a review. Livestock Science 114, 1–10.

Steinfeld, H., Gerber, P., Wassenaar, T., Castel, V., Rosales, M., De Haan, C., 2006. Livestock's long shadow: environmental issues and options. FAO, Rome, Italy.

Tamminga, S., Bannink, A., Dijkstra, J., Zom, R., 2007. Feeding strategies to reduce methane loss in dairy cattle. Report 34, Animal Sciences Group, Wageningen University, Wageningen, The Netherlands.

Thomassen, M.A., Dalgaard, R., Heijungs, R., De Boer, I.J.M., 2008a. Attributional and consequential LCA of milk production. The International Journal of Life Cycle Assessment 13, 339–349.

Thomassen, M.A., Van Calker, K.J., Smits, M.C.J., Iepema, G.L., De Boer, I.J.M., 2008b. Life cycle assessment of conventional and organic milk production in the Netherlands. Agricultural systems 96, 95–107.

Thomassen, M.A., Dolman, M.A., Van Calker, K.J., De Boer, I.J.M., 2009. Relating life cycle assessment indicators to gross value added for Dutch dairy farms. Ecological Economics 68, 2278–2284.

Udo, H.M.J., Cornelissen, T., 1998. Livestock in resource-poor farming systems. Outlook on Agriculture 27, 219–224.
USDA, 2009. National Nutrient Database for Standard Reference. http://www.nal.usda.gov/fnic/foodcomp/search.
Van Middelaar, C.E., Van Calker, K.J., Dolman, M.A., Berentsen, P.B.M., De Boer, I.J.M., 2009. Eco-efficiency of cheese production in the Netherlands.
Vellinga, T., Van Laar, H., Thomassen, M.A., De Boer, I.J.M., Berkhout, P., Aiking, H., 2008. Environmental impact of animal feed (Milieueffecten van diervoeders). Report 205, Animal Sciences Group, Wageningen University and research centre, Lelystad, The Netherlands.
Weidema, B.P., Wesnæs, M., Hermansen, J., Kristensen, T., Halberg, N., Eder, P., Delgado, L., 2008. Environmental improvement potentials of meat and dairy products. Report EUR 23491 EN. Institute for Prospective Technological Studies, Sevilla, Spain.
Williams, A.G., Audsley, E., Sandars, D.L., 2006. Determining the environmental burdens and resource use in the production of agricultural and horticultural commodities. Main Report Defra Research Project ISO205, Bedford: Cranfield University and Defra.
Zhu-XueQin, Van Ierland, E.C., 2004. Protein chains and environmental pressures: a comparison of pork and novel protein foods. Environmental Sciences 1 (3) 2003/2004.