Feed sources for livestock:

recycling towards a green planet

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Abstract

Livestock production has a major impact on the environment. Most of the impact of livestock production is related to feed production. To reduce the environmental impact, this thesis focused on using products for livestock feed that humans cannot or do not want to eat, such as co-products, food-waste, and biomass from marginal lands, further referred to as leftovers. This is an effective strategy, because it transforms an inedible stream into animal source food (ASF). We evaluated two mitigation strategies: replacing soybean meal (SBM) with rapeseed meal (RSM), and replacing SBM with waste-fed housefly larvae meal in pig diets. To assess the environmental benefits of these strategies, two methodological challenges had to be tackled first: how to include direct and indirect consequences of using leftovers as livestock feed in a life cycle assessment? And how to account for feed-food competition in a life cycle assessment (competition for land between humans and animals)? A consequential theoretical framework, therefore, was developed to account for indirect consequences. Solely based on the direct consequences, results showed that each mitigation strategy was promising (waste-fed larvae more so than RSM). Results were, however, contradictory when indirect consequences were included. Overall, including indirect consequences increased the environmental impact of each strategy. Especially the indirect consequences of feeding waste-fed larvae were large. This was because initially food-waste to feed larvae was used to produce bio-energy via anaerobic digestion. The environmental benefits related to replacing soybean meal with waste-fed larvae meal were less for global warming potential and energy use than environmental costs related to the marginal energy source, i.e. fossil-energy, replacing the bio-energy. Land use, nevertheless, was still largely reduced. The results, however, are situation specific: if the marginal energy source is wind or solar energy, the net environmental benefits of using larvae meal can be positive. Waste-fed larvae meal, therefore, appears to be an interesting mitigation strategy only when energy from wind and solar energy are used more dominantly than energy from fossil sources. To account for feed-food competition, a novel, holistic measure of land use efficiency, the so-called land use ratio (LUR) was developed. Results of the LUR showed that livestock production systems using mainly leftovers can produce human digestible protein more efficiently than crop production systems do. The availability of those leftover streams, however, is limited and, therefore, the amount of animal-source food (ASF) produced is also limited. The amount of ASF produced from livestock fed with only leftover streams was, therefore, also assessed. This results in a production 21 g of protein per person per day. On average, it is recommended to consume about 57 g of protein from ASF or plant-origin per person per day. Although ASF from default livestock does not fulfil the current global protein consumption of 32 g per person per day, about one third of the protein each person needs can be produced without competition for land between feed and food production. Livestock, therefore, can have an important contribution to the future nutrition supply. A paradigm shift, however, is needed in animal production to accomplish this: research in animal sciences should no longer focus on increasing efficiency of the animal or the animal production chain, but on increasing efficiency of the entire food system to ensure sustainable nutrition.

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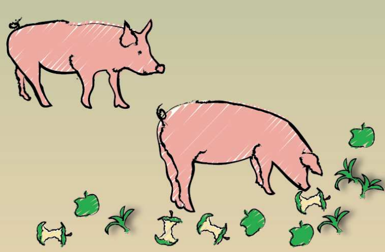

Chapter 1

General introduction

1 Background

The global human population has increased from about 3 billion people in 1961 to about 7 billion in 2011. A growing and wealthier population living increasingly in urban areas, demands an increase in housing, infrastructure, energy, and food (Thornton, 2010). Over the years, the global intake of animal-source food (ASF) increased from on average 20 g of protein per person per day in 1961 to 32 g of protein per person per day in 2011 (FAO, 2015). This increase in protein intake mostly involves ASF from monogastrics (Figure 1).

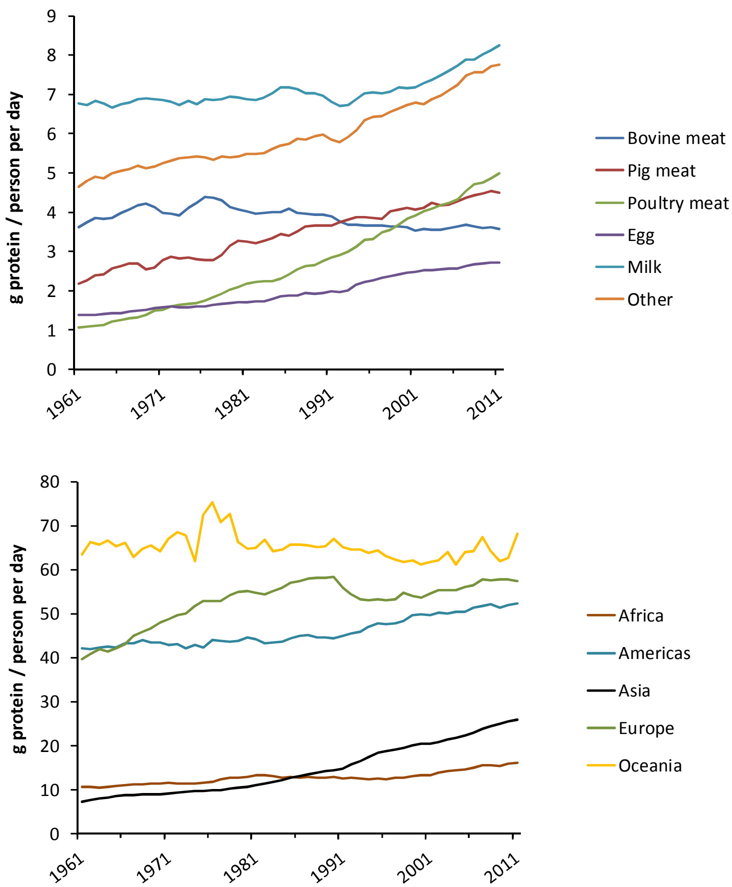

Figure 1. Consumption of gram protein per person per day by type of animal-source food and by world region. Data source, FAO 2015.

Note: Gram protein is used as a proxy to compare different types of ASF, because ASF is an important source of protein in human diets.

Consumption patterns of ASF, however, differ across global regions (Figure 1). Average consumption of ASF in Africa and Asia, for example, is still lower than in Europe, the Americas (North and South America) or Oceania. In 2011, a person from the United States consumed on average 71 g of animal-source protein per day, whereas a person from Europe consumed on average 61 g of animal-source protein and a person from Zambia consumed on average 9 g of animal-source protein (FAO, 2015). Consumption of ASF in developed countries increased until about 1990, after which it remained stable, whereas consumption of ASF in developing regions continues to increase (Thornton, 2010). Much of this increased consumption has been concentrated in countries that experienced rapid economic growth and evolved around poultry and pig production. The greatest increase in consumption of ASF has occurred in East and Southeast Asia (with the largest increase in China) and in Latin America and the Caribbean (with the largest increase in Brazil) (Thornton, 2010; Alexandratos and Bruinsma, 2012).

For an adult, the daily recommended intake of protein is approximately 57 g per person per day (EFSA, 2012), of which about one third is recommended to be from AFS (personal communication, Van 't Veer and Geleijnse, 2016) especially for some population groups, such as pregnant woman (Meier and Christen, 2012). In developed regions, such as Europe, consumption patterns are characterized by a high intake of animal protein, saturated fat, cholesterol, and calories (Hallström, 2015). This high intake can cause health issues, such as obesity, heart diseases, and cancer (Gerbens-Leenes et al., 2010; Micha et al., 2010; Kastner et al., 2012; Hallström, 2015). In developing regions, and for certain population groups (e.g. children), (increased) consumption of ASF is recommended to prevent health issues related to micronutrient deficiency, such as thiamine, vitamin B12 and Zn intakes (Temme et al., 2013; Temme et al., 2015).

In the event that no major changes occur, global predictions indicate that per capita consumption of cereals will stabilize, whereas meat consumption will increase by 76% and milk consumption will increase by 62% in the coming decade (Alexandratos and Bruinsma, 2012). These increases will occur mostly in developing regions, which, despite growing at a faster rate, will not reach a half of the consumption levels of the developed world by 2050 (Herrero et al., 2015). Altogether, increased demand for ASF has resulted, and is expected to continue to result in, accelerated growth of the livestock sector, or the so-called livestock revolution (Herrero et al., 2015).

The livestock revolution contributes to nutritional security in some parts of the world, generates economic benefits, results in improved livelihoods, and provides labour, but it also has drawbacks (Thornton, 2010). Livestock production causes severe environmental pressure via emissions to air, water, and soil (Steinfeld et al., 2006). The livestock sector is responsible for approximately 15% of the total anthropogenic emissions of greenhouse gases (Gerber et al., 2013) and scientific consensus links emissions of greenhouse gases from human activity to climate change (IPCC, 2007). Climate change has a negative impact on the environment and human health (Walther et al., 2002; McMichael et al., 2006), caused an increased rate of sea-level rise in the last decade (Church and White, 2006), increased bleaching and mortality in coral reefs (Stone, 2007) and increased the rate of large floods (Milly et al., 2002). Besides climate change, the livestock sector also has a large impact on eutrophication and acidification (Leip et al., 2015).

The livestock sector, furthermore, competes increasingly for scarce resources, such as land, water, phosphorus sources, and fossil-energy (Steinfeld et al., 2006; De Vries and De Boer, 2010; Leip et al., 2015). Global livestock production occupies about 30% of the permanent ice-free land on our planet, when all cropland and grassland used for feed are included (Steinfeld et al., 2006). In 2012, about 5 billion ha of global land was used for agriculture (FAO, 2015), of which about 70% was used for livestock production (Steinfeld et al., 2006). Of the 5 billion ha of agricultural land, about 1.6 billion ha is arable land, of which 33% is dedicated to feed-crops (Steinfeld et al., 2006). Increasing feed-crop production will lead to loss of grazing areas or deforestation, mainly in the tropics. For example, 80% of new croplands may replace forests, resulting in loss of biodiversity and increase of carbon emission (Foley et al., 2007; Gibbs et al., 2010; Foley et al., 2011).

One of the major challenges facing livestock production, therefore, is to contribute to global nutrition security in an environmentally sustainable way. Many proposed mitigation strategies for feeding the world sustainably focus primarily on reducing the impact of the livestock sector, so called production-side strategies (also called supply-side or efficiency strategies). Others focus on changing human diets, so called consumption-side strategies (also called demand restrained or sufficiency strategies) (Gerbens-Leenes and Nonhebel, 2002; Godfray et al., 2010; Foley et al., 2011; Garnett, 2011; Herrero and Thornton, 2013). Production-side strategies focus on increasing the production volume to meet the expected demand for ASF while increasing efficiency (decrease environmental impact per kg of ASF) and focus on technical innovations and managerial improvements (Garnett, 2014). Consumption-side strategies focus on changes in human diet patterns (Garnett, 2011; Smith et al., 2014; Schader et al., 2015). Eating less or no ASF is an often suggested solution to reduce the environmental impact of the human diet (Gerbens-Leenes and Nonhebel, 2002; Godfray et al., 2010; Foley et al., 2011; Garnett, 2011; Meier and Christen, 2012; Scarborough et al., 2014; Hallström et al., 2015). Furthermore, shifting the type of ASF, e.g. from ruminant meat to monogastric meat, is also often offered as a strategy to reduce the environmental impact of human diets (Wirsenius et al., 2010; Nijdam et al., 2012; Hallström et al., 2015).

Although both strategies (production- and consumption-side) are needed, this thesis primarily focusses on production-side strategies. The next paragraph will, therefore, focus on the environmental impact of the livestock sector and potential mitigation strategies.

2 The environmental impact of the livestock sector

Livestock production chains around the world are highly heterogeneous. They differ in agro-ecological environment e.g. climate variability; socio-economic situation, e.g. access to markets; use of external resources; and farm management, all of which result in the natural variability of livestock production systems. De Vries and De Boer (2010), furthermore, showed that there is a large variation in the environmental impact among livestock products. Compared to beef production, production of pork and chicken results in lower emissions of greenhouse gases, and requires less land and energy along the production chain than beef production (Figure 2). Differences in environmental impact among the production chain of pork, chicken, and beef can be explained in part by differences in feed efficiency, reproduction rate, and enteric methane (CH_4) emission between monogastrics and ruminants (Garnett, 2009; De Vries and De Boer, 2010).

Comparing the environmental impact of livestock production (Figure 3), feed production and utilization of feed has the largest impact on greenhouse gas (GHG) emissions and land use (LU) (De Vries and De Boer, 2010; Gerber et al., 2013). About half (47%) of all GHG emissions produced globally by the livestock sector are related to feed production (Gerber et al., 2013). There are no global studies showing the percentage of LU for the consumption of ASF, but several studies showed that per (regional) production systems the majority of LU originates from feed production (Basset-Mens and Van der Werf, 2005; Dalgaard et al., 2007; De Vries and De Boer, 2010).

Several strategies to reduce the impact of feed production can be identified. Sustainable intensification, for example, can be used to increase the yield on existing land, e.g. closing yield gaps (Godfray et al., 2010; Tilman et al., 2011). Another strategy is to improve feed efficiency, which has historically been the focus of the livestock sector and has been largely driven by economic incentives. Ruminants have a less efficient feed conversion compared to monogastrics (Šebek and Temme, 2009). This is based on a commonly used measure for feed efficiency, the feed conversion ratio (FCR). The FCR is defined as the amount of feed used per kg of animal product (kg feed intake/kg growth). The FCR is about 1.6 for broiler chickens, 2.5 for pigs, and 5.1 for cattle (Šebek and Temme, 2009). Decreasing the FCR will improve the feed efficiency of livestock systems.

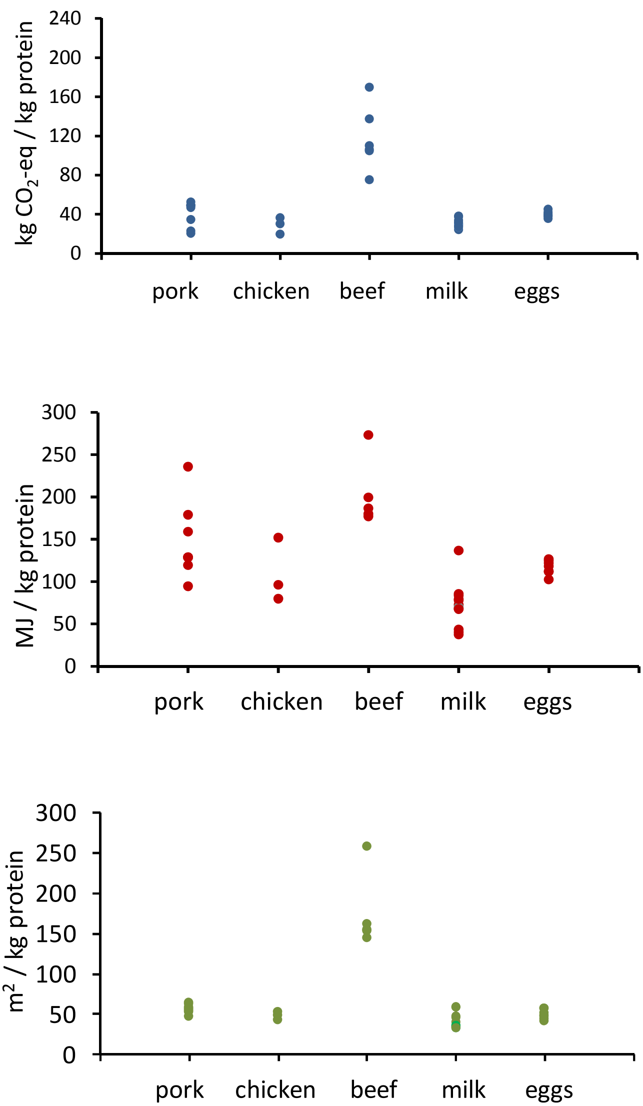

Figure 2. Global warming potential expressed in kg CO_2-eq; energy use expressed in MJ; and land use expressed in m^2 per kg of protein. Each point represents a study or scenario within a study (Adapted from De Vries and De Boer (2010)).

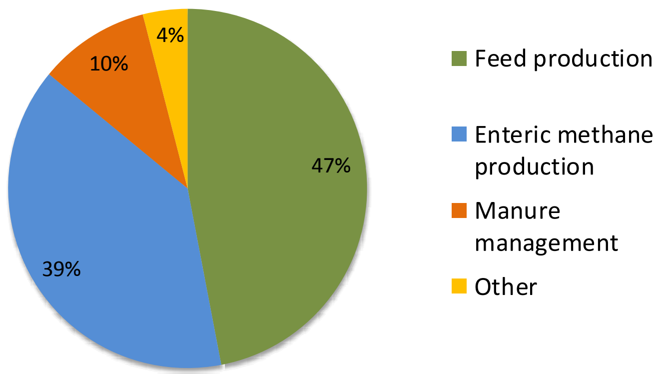

Figure 3. GHG emissions (%) related to various processes of the livestock sector.

The enduring focus on reducing FCR of livestock, however, has led to the use of large amounts of human-edible plant products, such as cereals, in livestock diets. Annually, about 1 billion tons of cereals are fed to livestock (Eisler et al., 2014). For future food security, it might be better not to use highly productive croplands to produce feed for livestock. No matter how efficiently cereals, pulses, and oilseeds are produced, direct consumption of these products by humans is ecologically more efficient than consumption of ASF produced by animals fed these products (Godfray et al., 2010; Foley et al., 2011). Feed produced on land suitable for human food production, therefore, results in competition for land between feed and food. The contribution of livestock production to nutrition security, in terms of maximizing the number of people to be nourished per ha, can be improved by feeding livestock mainly co-products from arable production or from food processing that are not edible by humans, or by grazing livestock on "marginal land", i.e. land not suitable for arable production (Garnett, 2009; Cassidy et al., 2013; Eisler et al., 2014).

Because an efficient use of non-edible human products and biomass from marginal land seems to play a significant role in the reduction of the environmental impact caused by the livestock sector (Godfray et al., 2010; Foley et al., 2011; Boland et al., 2013; Godfray and Garnett, 2014), the next paragraph focuses on this topic.

3 Leftover streams as livestock feed

There are several products that humans cannot or will not eat, but that are suitable as livestock feed, e.g. co-products, food-waste, and biomass from marginal land. Feeding co-products or food-waste to livestock or using biomass from marginal lands to feed livestock, further referred to as 'leftover streams', are effective options of using resources. By feeding leftovers, an inedible stream for humans can be transformed into high-quality food products, such as meat and milk (Nonhebel, 2004; Elferink et al., 2008; Garnett, 2009; Wirsenius et al., 2010).

Co-products are obtained throughout the harvesting or processing of human food. During the processing of sugar beet, for example, is not only sugar produced, but also beet-pulp and molasses. In such a multiple-output situation, a 'package of products' is produced. As sugar determines the production volume of sugar beets, sugar is defined as the 'determining product', and beet-pulp and molasses are defined as the 'co-products'. The production volume of a co-product, therefore, is driven by the demand for the determining product. If demand for sugar increases, production volume of sugar increases, which automatically increases production volume of beet pulp and molasses. If demand for beet-pulp increases, however, then production volume of beet-pulp will not increase, because volume of beet-pulp is determined by demand for sugar. If demand for beet-pulp increases in a way that beet-pulp becomes the product that economically drives the production process, then beet-pulp will become the determining product and sugar will become the co-product. In this thesis the term 'product-package' refers to the determining product and dependent co-products.

Products that are produced for human consumption, but that are wasted during retail or final consumption are referred to as food-waste in this thesis. The FAO estimated that, during production and consumption of food, about one third is wasted, and that per capita food-wasted by consumers in Europe and North America is about 95-115 kg per year (Gustavsson et al., 2011). High priority for strategies to reduce waste streams, therefore, seems logical. Part of our food-waste, however, is unavoidable. Feeding food-waste to livestock, therefore, can have a major contribution in reducing the environmental impact of the livestock sector (Boland et al., 2013; Zu Ermgassen et al., 2016). Food-waste has historically already been used as livestock feed, particularly for pigs (Zu Ermgassen et al., 2016). Pigs eat most foods also consumed by humans and can consume liquefied food, so they are an ideal target species to feed food-waste (Boland et al., 2013). Use of most food-waste as feed, however, is prohibited in many countries, including European countries, because of health and safety problems related to, for example, foot and mouth disease, African swine fever, and Bovine Spongiform Encephalopathy (BSE) (EC regulation 1774/2002). Intermediate innovations, such as feeding waste-fed insect to livestock, therefore, are being explored (Makkar et al., 2014; Van Huis, 2015). Although waste-fed insect are currently banned from livestock feed in the EU, we expect a rapid development of research initiatives and lobbying efforts to use waste-fed insects in the near future (Makkar et al., 2014; Van Huis, 2015). Using co-products and food-waste as livestock feed, therefore, would provide valuable nourishment for livestock and avoid feed-food[1] competition.

Besides co-products and food-waste, biomass from marginal lands can also be used to reduce the impact of the livestock sector on the environment. Marginal land includes areas that are less suitable or even unsuitable for crop production, because of rainfall, temperature or poor terrain limitations. Ruminants play an important role in grazing marginal land, because they can eat hay, silage, and high fibre crop residues that are unsuitable for consumption by humans and monogastrics (Fairly, 2010; Eisler et al., 2014). By doing so, ruminants relieve pressure on arable land, and retrieve otherwise inaccessible nutrients by adding them to the food chain (Fairly, 2010).

To conclude, considering the current environmental impact of the livestock sector and its challenge to meet the growing demand for ASF, the livestock sector is and will continue to be an important part of the puzzle to reduce global environmental impacts (Herrero and Thornton, 2013). Reduction of waste and efficient use of non-edible products are also recognised as key players in this transition (Godfray et al., 2010; Foley et al., 2011; Boland et al., 2013; Godfray and Garnett, 2014; Zu Ermgassen et al., 2016). It is essential, therefore, to explore mitigation strategies related to the use of leftovers in livestock diets.

4 Mitigation strategies to up-grade leftover streams

Examining the use of leftover streams, we can distinguish mitigation strategies on different system levels. A system is an entity that maintains its existence through the mutual interaction of it parts (Ten Napel et al., 2011). A system can be conceptualized at many levels: animal, farms, region, continent, and global (Figure 4).

This thesis focuses on three mitigation strategies. The first two mitigation strategies are production-side strategies and the last mitigation strategy combines production-side and consumption-side strategies, a so called consistency strategy. The mitigation strategies are explained below.

[1] 'Feed-food competition' in this thesis refers to the competition for resources such as land between crop production for human consumption and crop production for livestock feed.

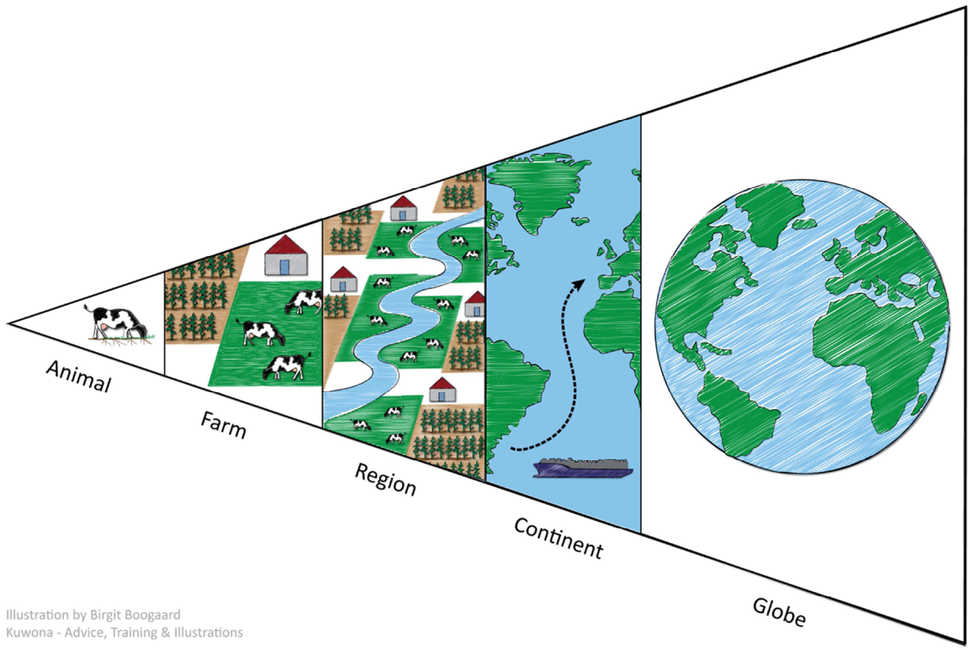

Figure 4. Conceptualisation of livestock systems on different levels.

4.1 Production-side strategy

Within production-side strategies two types of mitigation strategies can be distinguished: incremental innovations and system innovations. An innovation is a novel idea, practice, or product that significantly improves the environmental performance of a production system and can involve technologies, organisations, institutions, or policies (Asenso-Okyere and Davis, 2009).

An incremental innovation is a relatively small change or series of changes in an existing system (e.g. conventional pig farming) that lead to improvement (e.g. reduction of GHG emissions) in the production process of an existing product (e.g. pork). Incremental innovations are primarily characterised by changes in technology with relatively little alteration of the societal acceptance of these technologies (Elzen and Wieczorek, 2005).

A system innovation is an innovation that combines social, technical, and institutional change and, that interacts to result in the transformation of a system (Elzen et al., 2012). System innovations require technical as well as (long term) social and cultural changes (Elzen and Wieczorek, 2005).

Incremental innovation

Most studies that assess the environmental impact of the livestock sector focus on incremental innovations on farm level (e.g. De Vries and De Boer, 2010; Mosnier et al., 2011; Meul et al., 2012; Van Middelaar et al., 2014; De Vries et al., 2015). Three examples of incremental innovations related to the use of leftovers are: increasing the use of co-products in livestock diets, more efficient use of co-products by feeding them to those livestock species that convert most of the nutritional value of the feed, or replacing feed ingredients that have a high environmental impact with available novel feed ingredients. Currently, we see opportunities for using co-products from the bio-diesel and ethanol-fuel industry, e.g. rapeseed meal (RSM) and maize- and wheat dried distiller grain with soluble (DDGS). Co-products from bio-diesel production, especially RSM in the EU, became increasingly available during the last decade (Makkar et al., 2012). This increase is mainly due to the EU target to increase the use of bio-diesel in transport. Consequently, an increase in the average RSM content of livestock diets was seen, from 5% in 1994 to 12% in 2007 (Vellinga et al., 2009). In the Netherlands, RSM is mainly used in pig diets: in 2007, an average 3% in dairy cattle diets, 7% in poultry diets, and 12% in pig diets (Vellinga et al., 2009). Rapeseed meal is a protein-rich feed ingredient that can replace other protein-rich ingredients, such as soybean meal (SBM) (Thamsiriroj and Murphy, 2010; Reinhard and Zah, 2011). Cultivation of SBM has a high environmental impact, partly because of large transport distances and partly because nowadays SBM drives the production of soybean, which results in a high impact when based on economic allocation[2] (Cederberg and Flysjo, 2004; Van der Werf et al., 2005; Vellinga et al., 2009). Furthermore, SBM is related to land use change (LUC), such as deforestation in South America which results in release of CO_2 (Foley et al., 2007; Prudêncio da Silva et al., 2010). Because of SBM's high environmental impact, it is expected that replacing SBM with RSM will lead to a decrease in environmental impact, but this has never been investigated. The first innovation assessed in this thesis is, therefore, replacing SBM with RSM in livestock feed.

System innovation

Two examples of system innovations related to leftovers are: producing waste-fed insects for livestock feed (Makkar et al., 2014) and producing algae for bio-diesel production and the protein co-product for livestock feed (Craigie, 2011; Boland et al., 2013; Van der Burg et al., 2013). Research in the field of algae, however, is nascent so limited data are available. There

[2] Economic allocation is the partitioning of environmental impacts among co-products, based on the relative economic value of the outputs (Guinée et al., 2002).

are, however, studies of using waste-fed insects for livestock feed (Van Huis, 2013; Makkar et al., 2014; Van Huis, 2015).

Potential environmental benefits of rearing waste-fed insects for livestock feed suggest that insect-based feed might become an important alternative source of protein in the future (Van Huis et al., 2013; Makkar et al., 2014; Sánchez-Muros et al., 2014). Insects have a low FCR and can be consumed completely, without residual material such as bones or feathers. The nutritional value of insects is high, especially as a source of protein for livestock (Veldkamp et al., 2012; Makkar et al., 2014). In contrast to cultivation of feed crops, production of insects is not necessarily land intensive, in particular because insects can turn organic waste streams, such as manure or food-waste, into high-quality insect-based feed products (Veldkamp et al., 2012; Van Huis et al., 2013; Sánchez-Muros et al., 2014). From an environmental perspective, however, it is more efficient to use food-waste directly as livestock feed, nevertheless, feeding waste-fed insects might be an intermediate innovation to up-grade food-waste streams. A study that focussed on production of mealworms for human consumption, showed that compared to protein from livestock, the production of one kg of edible protein from mealworms resulted in less LU, but a higher global warming potential (GWP), and higher energy use (EU) (Oonincx and De Boer, 2012). It is questionable, therefore, whether or not the production of waste-fed insects will result in environmental benefits. The second innovation assessed in this thesis is, therefore, the use of waste-fed insects as livestock feed.

4.2 Consistency strategy

The consistency strategy combines production-side strategies and consumption-side strategies (Huber, 2000; Schader et al., 2015). Based on the principles of the consistency strategy, livestock systems should mainly and optimally use co-products, food-waste, and biomass from marginal land in livestock feed, to minimise use of human-edible feed ingredients. Livestock that eat only leftovers do not compete with humans for cropland, and also contribute to sustainable nutrition security. Such a transition[3] requires a change in focus from increasing productivity per animal towards increasing the number of people to be nourished per hectare (Cassidy et al., 2013). This change in focus means making optimal use of leftovers. However, when livestock are only fed with leftover streams, less ASF can be produced. A strategy based on feeding only leftovers, therefore, requires changes not only on the production-side but also on the consumption-side. One can wonder, however, how much

[3] The term ‘transition’ highlights a difference between an earlier and a later stage of livestock production (e.g. horse-power based versus tractor-power based) (Elzen et al., 2012).

ASF can we consume by feeding leftover steams to livestock? The last innovation assessed in this thesis, therefore, is the sole use of leftovers as livestock feed.

5 Methodological challenges

To assess the environmental impact of the above mentioned innovations (replacing SBM with RSM or waste-fed insects and feeding only leftovers) a life cycle assessment (LCA) is generally used. LCA is an internationally accepted and standardized method (ISO14040, 1997; ISO14041, 1998; ISO14042, 2000; ISO14043, 2000) to evaluate the environmental impact during the entire production system (Guinée et al., 2002; Bauman and Tillman, 2004). An LCA assesses the impact from the extraction of raw materials, via production, and processing, via packaging, transport, and up to product use and waste disposal (Figure 5). Although LCA methods are generally used to assess the environmental impact of ASF, two questions occur when the impact of the mitigation strategy are applied: How to deal with product-packages? and How to account for feed-food competition?

Figure 5. Processes of a life cycle assessment, from the extraction of raw materials, via production, processing, transport, and via packaging, to product use, and waste disposal.

5.1 Product-packages

There are two LCA methods: attributional LCA (ALCA) and consequential LCA (CLCA), each using a different way to deal with product-packages (Ekvall and Weidema, 2004).

To assess the environmental impact of the livestock sector, the commonly used ALCA can be used. An ALCA assesses systems in a status quo situation, and, therefore, describes the environmentally relevant physical flows to and from a product or system (Bauman and Tillman, 2004). For each process of the system, the related environmental impact is determined and the impact of all processes is summed, resulting in the environmental impact of the system. For multifunctional processes, economic allocation is commonly used in ALCA studies of livestock products (De Vries and De Boer, 2010).

By using economic allocation, however, the complexity of dealing with product-packages is not fully grasped. Economic allocation does not account for the dependency of the co-product related to the determining products (e.g. the production volume of beet-pulp depends on the demand for sugar). Furthermore, leftover streams can already have applications, resulting in the shifting of the application of the product from one sector to another sector. Whether or not this results in an improved net environmental impact depends on the environmental benefits of using the product in its new application minus the environmental cost of replacing the product in its old application. Food-waste, for example, can be used not only as feed for insects replacing conventional feed ingredients, but can also be used for the production of bio-energy. The environmental impact of the production of energy needed to replace the original bio-energy function of food-waste, must be accounted for. This is one example of an indirect consequence of using waste-fed insects for livestock feed. Such indirect consequences need to be considered when evaluating the environmental impact of innovations.

To assess such indirect consequences, a CLCA can be used. A CLCA describes how environmental flows/processes change within and outside the production cycle of a product, in response to a change in the system (Ekvall and Weidema, 2004). Performing a CLCA predicts the consequences of an action, and, therefore, requires insight into cause-and-effect chains (Ekvall and Weidema, 2004). Current CLCA methods, as described by Weidema et al. (2009) focus on assessing the environmental impact of an increased demand of the determining product. A framework to determine the environmental impact of using leftovers in livestock feed, however, is lacking.

5.2 Feed-food competition

Another methodological challenge is calculating land use efficiency. LCA studies focus on the total amount of land required to produce one kg ASF, and that includes plant and animal productivity. Interpretation of LCA studies related to land use efficiency, however, are hindered because results do not include differences in consumption of human-edible products by different livestock species or differences in suitability of land used for feed production to directly cultivate food-crops. In other words, LCA studies do not account for competition for land between humans and animals. Given the global constraints on land we should grow food directly for human consumption rather than for livestock (Nonhebel, 2004; Garnett, 2009; Godfray et al., 2010; Foley et al., 2011). To address the contribution of livestock to the future food supply, a measure for land use efficiency is needed that accounts for plant productivity, efficiency of converting human-inedible feed into ASF, the suitability of land for crop cultivation, and has a life-cycle perspective.

6 Aim

The aims of this study were to:

- Develop theoretical frameworks that enable evaluation of environmental consequences of using leftovers as livestock feed, while accounting for product-packages and feed-food competition.
- Assess the environmental impact of innovations related leftovers, by using these theoretical frameworks.

7 Outline of the thesis

The structure of this thesis is shown in Figure 6. In this thesis, emissions of greenhouse gases (GHGs), EU, and LU were assessed. Emission of GHGs and LU were chosen as examples, as the livestock sector contributes significantly to both climate change and LU worldwide (Steinfeld et al., 2006). Furthermore, EU was used because it influences GWP and plays a role in, for example, the rearing of insects. In the Chapters related to the use of co-products and food-waste, pigs were used as an example (Boland et al., 2013; Zu Ermgassen et al., 2016), whereas in Chapters related to the use of marginal land use (dairy) cattle were used as an example (Fairly, 2010; Eisler et al., 2014).

In Chapter 2, a sensitivity analysis was performed for the pork production chain on emissions of GHGs, to identify the effect of each input parameter on a model output. LU was not assessed, as results were expected to be straightforward because the majority of land used is caused by feed production. The input parameters that had a large effect on the output were carefully considered when the environmental impact of mitigation strategies related to the use of leftovers were assessed.

In Chapter 3 the environmental impact of replacing SBM with RSM in pig diets was assessed based on an ALCA. In Chapter 4, a theoretical framework, based on a CLCA was developed to assess the environmental impact of replacing one conventional feed ingredient with one leftover feed ingredient (on feed ingredient level e.g. wheat middlings or molasses). In Chapter 5, the environmental impact of producing waste-fed housefly larvae as livestock feed was assessed, based on the theoretical framework developed in Chapter 4. In Chapter 6, the theoretical framework developed in Chapter 4 was extended to pig diet level. To illustrate the framework on pig diet level, the innovations of Chapter 3 (replacing SBM with RSM) and Chapter 5 (producing waste-fed larvae) were applied. In Chapter 7, a theoretical framework was developed to assess efficiency of land use including feed-food competition. The framework was illustrated with three case studies based on two Dutch production systems (dairy cattle and laying hens). In Chapter 8, the amount of ASF that can be consumed was assessed, when feed-food competition was avoided by feeding co-products and food-waste to pigs and by using biomass from marginal land to feed ruminants.

The first objective of this thesis, related to theoretical frameworks, was addressed in Chapters 4, 6, and 7. The second objective of this thesis, related to innovations was addressed in Chapters 3, 5, and 6.

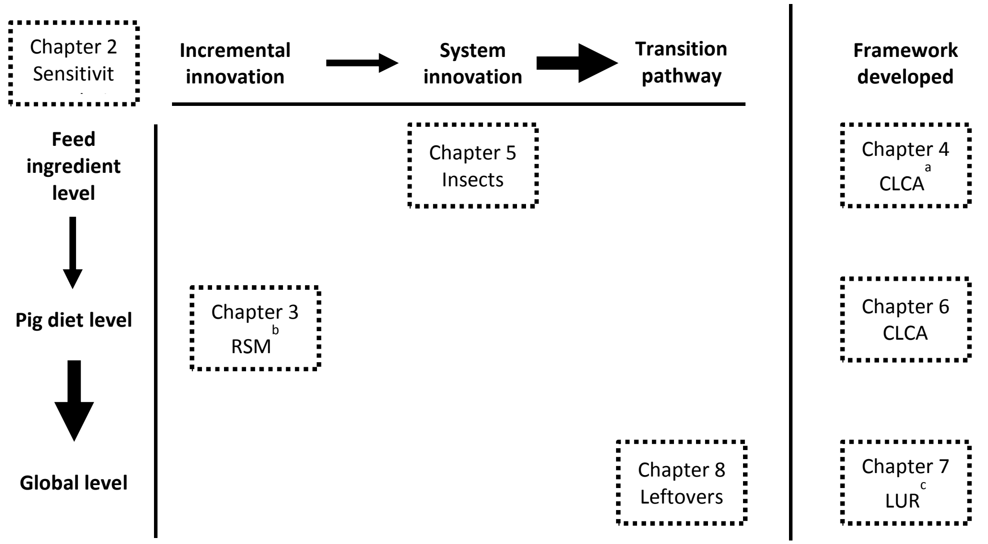

Figure 6. Structure of this thesis.
[a] consequential life cycles assessment, [b] rapeseed meal, and [c] land use ratio

Chapter 2: *Sensitivity analysis of greenhouse gas emissions of a pork-production chain*

This study aimed to identify the most important input parameters in an LCA that contribute to greenhouse gas emissions of a pork-production chain. Based on the results of this study, we identified the effect of each input parameter on a model output.

Chapter 3 : *Environmental impact of replacing soybean meal with rapeseed meal in diets of finishing-pigs*

This study aimed to assess the impact of replacing SBM with RSM on GWP, EU, and LU along the entire pig-production chain. Using a sensitivity analysis, we explored the impact of including emissions from direct and indirect LUC, changing parameters to characterize pig growth, and using various methods to calculate emissions from manure management.

Chapter 4: *Assessing environmental consequences of using co-products in animal feed*

This study aimed to develop a theoretical framework, based on CLCA principles, that assists in how to assess the environmental impact (GWP and LU) of using co-products. We focus on increasing the use of co-products within the livestock sector.

Chapter 5: *From environmental nuisance to environmental opportunity: housefly larvae convert waste to livestock feed*

This study aimed to explore whether the environmental impact (GWP, EU, and LU) of livestock production can be reduced by using larvae of the common housefly, grown on organic waste streams, as livestock feed. To assess the environmental impact both an ALCA and a CLCA were performed.

Chapter 6: *Consequential life cycle assessment and feed optimization: alternative protein sources in pig diets*

This study aimed to explore differences in environmental impact (GWP, EU, and LU) in using an ALCA or a CLCA on pig diet level. Three scenarios were used: a conventional pig feed including SBM, a pig feed in which SBM was replaced with RSM, and finally a pig feed in which SBM was replaced with waste-fed larvae meal.

Chapter 7: *Global food supply: land use efficiency of livestock systems*

This study aimed to develop a method to assess land use efficiency, including feed-food competition. Land use ratio (LUR) was defined as the ratio of the maximum amount of human digestible protein (HDP), derived potentially from food-crops on all land used to cultivate feed, required to produce one kg ASF to the amount of HDP in that one kg ASF.

Chapter 8: *The role of livestock in a sustainable diet: a land use perspective (Opinion paper)*

This study aimed to assess how much ASF can be consumed when feed-food competition was avoided by feeding co-products and food-waste to pigs and by using biomass from marginal land to feed ruminants.

Chapter 2

Sensitivity analysis of greenhouse gas emissions from a pork production chain

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Abstract

This study aimed to identify the most essential input parameters in the assessment of greenhouse gas (GHG) emissions along the pork production chain. We identified most essential input parameters by combining two sensitivity-analysis methods: the multiplier method and the method of elementary effects. The former shows how much an input parameter influences assessment of GHG emissions, whereas the latter shows the importance of input parameters on uncertainty in the output. For the method of elementary effects, uncertainty ranges were implemented only for input parameters that were identified as being most influential based on the multiplier method or that had large uncertainty ranges based on the literature. Results showed that the most essential input parameters are the feed-conversion ratio, the amount of manure, CH_4 emissions from manure management and crop yields, especially of maize and barley. Combining the results of both methods allowed derivation of mitigation options, either based on innovations (e.g. novel feeding strategies) or on management strategies (e.g. reducing mortality rate), and formulation of options for improving reliability of the results. Mitigation options based on innovations were shown to be most effective when directed at improving the feed-conversion ratio; decreasing the amount of manure produced by pigs; improving maize, barley and wheat yields; decreasing the number of sows or piglets per growing-pig needed and improving efficiency of N-fertiliser production. Mitigation options based on management strategies were shown to be most effective when farmers strive to reduce feed intake, reduce application of N fertiliser to maize and barley, and reduce the number of sows per growing-pig needed towards best practices. Finally, the method of elementary effects showed that reliability of assessing GHG emissions of pork production could be improved when uncertainty ranges are reduced, for example, around direct and indirect N_2O emissions of the main feed crops in the pig diet and the CH_4 emissions of manure. Also the reliability could be improved by improving data quality of the most essential parameters. Combining two types of sensitivity-analysis methods identified the most essential input parameters in the pork production chain. With this combined analysis, mitigation options via innovations and management strategies were derived, and parameters were identified that improved reliability of the results.

1 Introduction

Environmental impacts of the agri-food industry have been of increasing concern; in particular, awareness about environmental impacts of animal production are increasingly acknowledged (Steinfeld et al., 2006). The livestock sector, for example, is responsible for about 15% of the total anthropogenic emissions of greenhouse gases (Gerber et al., 2013). Worldwide, pork production explains about 9% of greenhouse gas (GHG) emissions of the livestock sector (Gerber et al., 2013). In general, the environmental impact of pork production is quantified using life cycle assessment (LCA) (Bauman and Tillman, 2004). To quantify GHG emissions of the entire pork production chain, we need to define values for input parameters, such as feed-conversion ratios, crop yields, nitrogen application ratios, and emission factors. Uncertainty around these input values can cause a large variation in GHG emissions estimates. For example, within the IPCC tier 1 framework, direct N_2O emissions of N from fertiliser and manure and crop residues vary by a factor of ten: 0.003 to 0.03 kg N_2O per kg N applied (IPCC, 2006).

To quantify to what extent environmental impacts of pork production chain varied and to explore the robustness of the results, Basset-Mens and Van der Werf (2005), Basset-Mens et al. (2006), and Van der Werf et al. (2005) identified ranges of some of their input parameters and assessed the effect of these ranges on the output. Basset-Mens and Van der Werf (2005) for example, concluded that N_2O emissions of feed crops caused large uncertainty around estimates of total GHG emissions, indicating that the impact of the feed crops is high, as are the uncertainty ranges around their emissions. None of these studies systematically explored the effect, or contribution, of each individual input parameter to the output. However, it is possible to assess the importance of each individual parameter in an LCA model by performing a sensitivity analysis.

Most LCA studies that performed a sensitivity analysis used a straightforward method, i.e. a one-at-a-time (OAT) approach. An OAT approach selects an input parameter and changes it e.g. 10%, and subsequently quantifies the effect on model output (Suh and Yee 2011; Van Middelaar et al., 2013; Van Zanten et al., 2015a; Yang et al., 2011). By exploring the impact of input parameters on the output, the robustness of the results is explored. The input parameters that cause most change in model output are considered to be the most *influential parameters*. The OAT approach is often chosen because of its simplicity as it is not necessary to gather additional data or to derive, for example, ranges or distribution functions for all input parameters (Bjorklund, 2002). However, the OAT approach has two weaknesses. First, the number of input parameters assessed is usually a subset of all input parameters, implying that potential influential parameters might be overlooked. Second, the arbitrary choice of

10% may not reflect the actual uncertainty range of the input data. Some input parameters may vary only 5%, while others may vary by a factor of ten. Therefore, the actual effect on the output might be under- or overestimated.

Two methods for sensitivity analysis are available that overcome these weaknesses. The multiplier method (MPM) determines the influence of all input parameters in an LCA model, and, therefore, accommodates the first weakness. MPM was first introduced in LCA by Heijungs (1994) but to our knowledge has not been applied to an agricultural case study in LCA. MPM can be used to determine areas of potential mitigation options (Heijungs, 1996) but does not take into account the actual ranges over which the input parameters can vary. In contrast, the method of elementary effects (MEE) does include an uncertainty range for each input parameter, and, therefore, accommodates the second weakness mentioned. MEE calculates the importance of the input parameters based on their actual ranges, by exploring model outputs within these ranges. MEE can be used to determine how much the uncertainty around the input parameters affects the output. The parameters that affect the output most, based on their uncertainty range, are referred to as the most *important parameters*. It should be noted that although MEE provides a sampled model output, it is primarily used for sensitivity analysis belonging to the area of screening methods (Saltelli et al., 2008). MEE was originally designed by Morris (1991) and expanded by Campolongo et al. (2007). To our knowledge, MEE has only been applied to LCA studies outside livestock production e.g. cocoa production by Mutel et al. (2013) and detergent production by de Koning et al. (2010).

This study aims to identify the most *essential parameters* in an LCA model of GHG emissions of pork production by combining results of the two sensitivity-analysis methods. First, MPM is applied, including all input parameters in the model, and second MEE is applied, which explores consequences of actual ranges in uncertainty. Combining results of both methods may help to formulate potential mitigation options and increase reliability of LCA results.

2 Material and methods

2.1 Matrix formulation in LCA

To facilitate the use of the sensitivity-analysis methods applied in this study, we used matrix-based LCA (Heijungs and Suh, 2002). The inventory totals equal:

$$\mathbf{g} = \mathbf{B}\mathbf{A}^{-1}\mathbf{f} \qquad (1)$$

Input parameters of an LCA consist of technical parameters and emissions or resource use. The technology matrix **A** contains the technical parameters of various production processes included in the chain, such as production of feed or storage of manure, presented as a set of linear equations. Each column represents a production process. The associated emissions are found in the **B**-matrix, e.g. the kg CH_4 per kg manure storage per year. The **A**-matrix is scaled to produce the amount given by the functional unit **f** (e.g. kg of growing-pig). To calculate the total environmental impact per impact category (**h**), the inventory result (**g**) is multiplied by the characterisation matrix (**Q**):

$$\mathbf{h} = \mathbf{Qg} \tag{2}$$

In this case, **Q** contains the characterization factors of GHG emissions for global warming potential (GWP) on a 100-year time interval: carbon dioxide (CO_2), biogenic methane ($CH_{4,\ bio}$): 28 kg CO_2-eq/kg biogenic methane, fossil methane ($CH_{4,\ fossil}$): 30 kg CO_2-eq/kg fossil methane; and nitrous oxide (N_2O): 265 kg CO_2-eq/kg nitrous oxide (Myhre et al., 2013), thus reducing to a vector **q'** and **h** to a scalar h. All modelling in this paper was performed in MATLAB, and the code is available upon request to the authors. We only considered elements in **A** and in **B** to contain uncertainty; **f** and **Q** remained fixed.

2.2 Multiplier method

MPM predicts the change in the result h of a small change around the default value of each input parameter in **A** or **B**. A derivation of the method can be found in Heijungs (2010). MPM uses first-order partial derivatives $\left(\frac{\partial(h,m)}{\partial(A,i,j)}\right)$ and $\left(\frac{\partial(h,m)}{\partial(B,i,j)}\right)$ to estimate the influence around each input parameter. To compare the influence of the input parameters, the partial derivatives are normalized with respect to their original value A_{ij} and B_{kj}, where A_{ij} and B_{kj} are elements of **A** and **B** respectively, and h_m are the impact categories in h. The multipliers equal:

$$\eta(h,m;A,i,j) = \frac{A_{ij}}{h_m}\frac{\partial(h,m)}{\partial(A,i,j)} \tag{3}$$

$$\eta(h,m;B,k,j) = \frac{B_{kj}}{h_m}\frac{\partial(h,m)}{\partial(B,kj)} \tag{4}$$

Full expressions of the multipliers of equation (3) and (4) are given in Heijungs (2010). The multiplier will give not only the magnitude but also the direction of change, and can either be positive or negative. The multipliers can be interpreted as how much a 1% change in the input will affect the output (in %). For illustrational purposes, we will also use the absolute effect, given by $|\eta|$.

2.3 Method of elementary effects

MEE uses the actual ranges of each input parameter. A range is defined as a minimum and a maximum for each parameter, and can originate from variability or epistemic uncertainty. Variability in input parameters arises from e.g. variation in crop yields or N-fertiliser rates over years; it is inherent to the data and cannot be reduced. Epistemic uncertainty comes from unknowns around an input parameter (Walker et al., 2003), and is for example found for the IPCC emission factors of N_2O emissions of fertiliser application. Gaining more knowledge about an input parameter, e.g. by better measurements, can reduce epistemic uncertainty (Chen and Corson, 2014).

The minimum and maximum value for each input parameter can be used to calculate the combination of all minima and maxima, but in the case of 100 parameters, this would lead to $2^{100} \approx 10^{30}$ calculations. This approach may not be feasible, especially for large models. To overcome this problem, MEE selects two points within the range for each input parameter and calculates the change in the output based on this change in the input parameter, changing each parameter only once. To perform MEE, the range of each input parameter is divided into three equal parts (it does not use the default value, as MPM does). If a parameter ranges from 0 to 1, for example, the division would lead to (0; 1/3); (1/3; 2/3); (2/3; 1). It is possible to create smaller or larger divisions, but this is a common choice (Campolongo et al., 2007; Saltelli et al., 2008). Starting from an arbitrary starting point, one parameter is selected at random and changed with a predefined step size $\delta_{A,i,j}$ or $\delta_{B,k,j}$, set to $2/3$ of the range of each input parameter (Campolongo et al., 2007). This is repeated until each parameter has changed once (one trajectory has been performed), and the elementary effects ($EE(A,i,j)$) and ($EE(B,k,j)$) can be calculated for each parameter A_{ij} and B_{kj} by dividing the change in output by the step size Δ (equal to $2/3$):

$$EE(A,i,j) = \frac{h(A_{ij}+\delta_{A,i,j}) - h(A_{ij})}{\Delta} \quad (5)$$

$$EE(B,k,j) = \frac{h(B_{kj}+\delta_{B,k,j}) - h(B_{kj})}{\Delta} \quad (6)$$

The above procedure is repeated several times. A measure of importance is found by calculating μ*, which is the (absolute) mean of the average elementary effects[1]:

[1] As the trajectories are chosen at random, one can imagine that the choice of the trajectories can be closer or further apart. In an optimal situation, one would like the trajectories to be as far apart as possible. Campolongo et al. (2007) proposed a brute force approach that selects e.g. ten more optimal trajectories from a set of 100. As this model is linear, e.g. the ranking of the parameters did not change for multiple runs, we did not include this part of the method, as it takes much more computational effort in terms of run-time and memory usage.

$$\mu^*(A,i,j) = \frac{1}{R}\sum_{r=1}^{R}|EE(A,i,j)| \text{and} \qquad \mu^*(B,k,j) = \frac{1}{R}\sum_{r=1}^{R}|EE(B,k,j)| \tag{7}$$

where R is the number of trajectories (usually set to 10). The set of μ^* values can be ranked from the most to the least important parameter[2].

2.4 Framework for combining MPM and MEE

We combined MPM and MEE based on a figure in Heijungs (1996), which distinguished between the *influence* and the *importance* of input parameters on output uncertainty. If an input parameter is both influential and important, the parameter is considered as *essential* (Figure 1). We used MPM to determine the influence and MEE to determine the importance of each input parameter.

We adapted the figure from Heijungs (1996) to identify mitigation options based on innovations or management strategies (Figure 1). The horizontal axis ranks the most influential parameters, which, therefore, could have most impact if they are reduced. These mitigation options reflect innovations in the production chain. The vertical axis ranks the parameters that are most important to output uncertainty, caused by either variability due to e.g. differences in management practises, or epistemic uncertainties. Input parameters that are highly important and highly influential can be used to identify potential mitigation strategies (i.e. essential parameters, Figure 1, top right corner).

Environmental impacts of the livestock sector, for example, can be reduced if farmers adapt their management strategies towards those farmers, with a relatively low environmental impact. In addition, reliability can be improved by reducing the epistemic uncertainties that are shown to be important, which are also found in the direction of the vertical axis. Reducing the epistemic uncertainty ranges of input parameters that affect the output highly would lead to smaller ranges of uncertainty around the output, hence more reliable conclusions. Epistemic uncertainties can be reduced by better measurements. Reliability can also be improved by improving data quality of the most essential parameters.

[2] Another indicator that can be calculated is σ, which is an indicator of interaction or non-linear effects: if the elementary effect of a certain parameter changes for different trajectories, the magnitude of the elementary effect depends on either the configuration of the model or the presence of nonlinear effects, but this will not be discussed in this article.

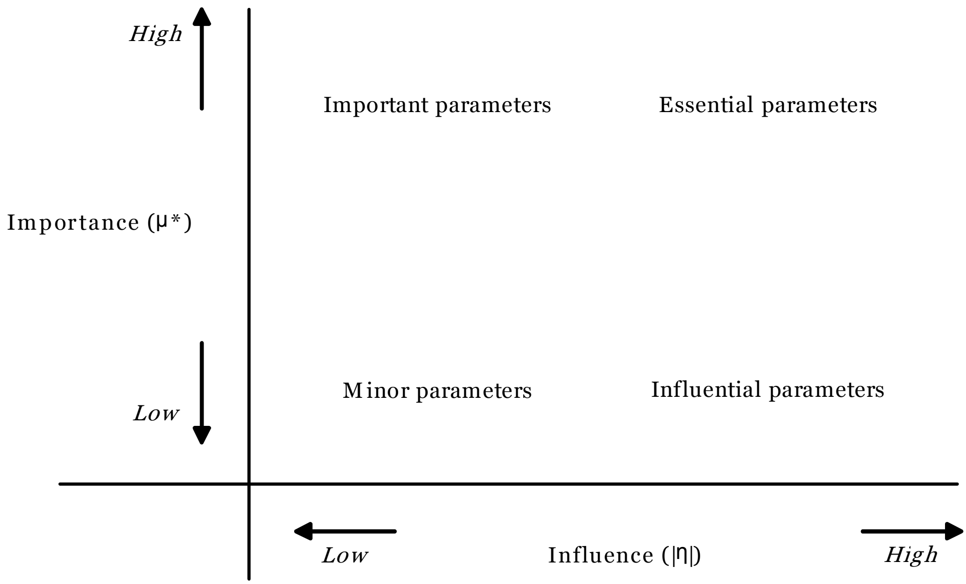

Figure 1. Framework for combining MPM and MEE. The most influential and important parameters are shown in the top right corner (essential parameters). Adapted from Heijungs (1996).

2.5 Case study: pork production chain

Pork production system

The pig production model is based mainly on van Zanten et al. (2015b), and the functional unit is one kg body weight of a growing-pig. Environmental impacts of the following processes in the pig chain were considered and are explained below: production of crop inputs (e.g. fertiliser), feed processing (e.g. milling), piglet production (rearing), manure management, pig housing, and enteric fermentation from pigs (Figure 2). The Appendix (Table A.1) provides the compositions of diets for growing-pigs, piglets, gilts, and sows. Diet compositions were an average representation for 2012 and were composed based on the procedure described by Bikker et al. (2011). The average diet contained four diets, one for each quarter of the year. The diets were formulated using a commercial linear programming tool for feed (i.e. Bestmix®, Adifo, Maldegem, Belgium), which optimises a diet by minimising the cost of the diet (Nuscience 2012). The diets had to meet the average nutritional requirements for the pigs in Dutch practice, e.g. growing-pig diets contained 9.68 MJ net energy per kg feed. To assess the average growth performance (aligned with the nutritional content of the diet), annual average company data of Dutch pig farms were used

(Agrovision, 2012). Piglets had a start weight of 25 kg. After 118 days, growing-pigs were ready for slaughter, weighing 118 kg on average. In case of a multifunctional process (e.g. production of soybean oil and soybean meal), economic allocation was used, which is the partitioning of environmental impacts between co-products based on the relative economic value of the outputs (Guinée et al., 2002). Economic allocation is used most commonly in LCA studies of livestock products (De Vries and De Boer, 2010).

Inventory

Data were collected for each of the stages in the production process of pork (Figure 2): (1) production of crop inputs; (2) crop cultivation, including transportation and processing; (3) feed production; (4) pork production; (5) manure management; (6) enteric fermentation and (7) housing of the pigs (Appendix, Table A.2). For MPM, all default data can be found in the Appendix. For MEE, we tried but were unable to determine ranges for all input parameters. Based on the literature (e.g. uncertainty ranges around direct N_2O emission factors) and our own analysis with MPM (section 3.2), we identified the most important parameters to be included. Input parameters (technical parameters and emissions) for which we could quantify uncertainty ranges are discussed in more detail below (Tables 1 to 5).

The pork production chain contained 354 input parameters; all were analysed in MPM, and 46 were considered in MEE. Ranges for MEE were based both on variability in farm data and epistemic uncertainties around the input parameters.

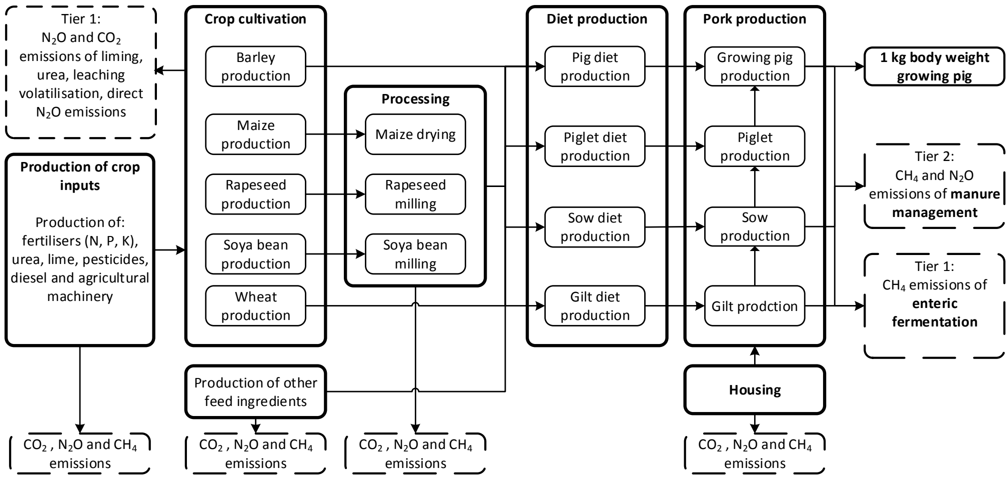

Figure 2. Production processes in the pork production chain; solid boxes are production processes, while dotted boxes refer to emissions.

Production of crop inputs and crop cultivation

Diets of growing-pigs, piglets, sows, and gilts consisted of 31 ingredients in total (Appendix, Table A.1) and represented a mean feed intake of 244 kg, ranging from 234-352 kg (Table 1), including feed intake related to mortality of growing-pigs (Table A.2). Uncertainty ranges were assumed only for the five ingredients that contributed most to GHG emissions in the diet of the growing-pig, identified with the MPM method. These five ingredients were barley, maize, rapeseed (meal), soybean (meal) and wheat. Data on feed processing and feed transportation are given in the Appendix (Table A.3).

Table 1. Default values of the feed composition of five ingredients that were largest in mass-share for pig, gilt, sows and piglets, including a range in the total feed intake of the growing-pig.

	Pig production	Gilt production	Sow production	Piglet production
Feed intake (kg)	244 (234 - 352)[a,b]	403[c]	1174[d]	29[e]
Barley (%)	12.9	6.78	13.7	32.1
Maize (%)	26.7	25.0	21.1	21.5
Rapeseed meal (%)	10.2	10.0	1.30	n/a
Soybean meal (%)	7.50	4.25	3.70	12.9
Wheat (%)	20.9	20.4	12.4	11.0
Other (%)	21.8	33.6	47.8	22.5

[a] Personal communication M, Dolman, LEI Wageningen UR (May 11, 2015)
[b] kg feed per growing-pig
[c] kg feed per gilt
[d] kg feed per sow per year
[e] kg feed per piglet

Table 2. Default values and ranges of yields and fertiliser application rates per crop type per year.

Ingredient	Yield (kg DM[g]/ha) Default[a]	Range[b]	N-fertiliser application rate (kg N/ha) Default[a]	Range[c, d]
Barley	5520	4828-5809	130	76-130
Maize	7621	5917-7621	150	64–294
Rapeseed	3040	2800-3477	73.4	49-78[(e)]
Soybean	4800	4342-5099	9	0-12[(f)]
Wheat	6010	5245-6451	55	43-64[(e)]

[a] Default: Garcia-Launay et al. (2014)
[b] Range: FAOSTAT (based on 5 years: 2009-2013) for France (barley, maize, rapeseed, wheat) and Brazil (soybean)
[c] Minimum: Basset-Mens and Van der Werf (2005) (red label; solid manure for barley, maize and wheat converted to N)
[d] Maximum: Meul et al. (2012), except for barley: Garcia-Launay et al. (2014)
[e] Adjusted ranges for N urea and N fertiliser
[f] For two harvests per year
[g] DM: dry matter

Default values and ranges for yields and N-fertiliser applications for the five main ingredients in the pig diet were defined (Table 2). Ranges are caused by (natural) variability around the input parameters. The default data of the technical parameters that were fixed, e.g. inputs for crop production, are given in in the Appendix (Table A.3).

Direct and indirect N_2O and CO_2 emissions due to liming and urea application were quantified, including their ranges according to IPCC tier 1 (Appendix, equations A.1 – A.9). The ranges are caused by epistemic uncertainties around the emission factors (IPCC, 2006). For the other feed ingredients, default values were included (Vellinga et al., 2013). CO_2 emission factors from urea application and liming, direct and indirect N_2O emission factors of the five main ingredients (barley, maize, rapeseed, soybean, and wheat), and their ranges, were also included (Table 3).

Table 3. Emission factors for CO_2 emissions and direct/indirect N_2O emissions based on IPCC tier 1 per crop per year.

Emission factor	Default	Range
CO_2 from liming (kg CO_2-C/(kg·yr))	0.12	0.06-0.12
CO_2 from urea (kg CO_2-C/(kg·yr))	0.2	0.1-0.2
Direct N_2O (kg N_2O-N/(kg·yr))	0.01	0.003-0.03
Indirect N_2O from leaching (kg N_2O-N)/(kg·yr))	0.0075	0.005-0.025
Indirect N_2O from volatilisation (kg N_2O-N/(kg·yr))	0.01	0.002-0.05

Table 4. Emission factors for direct and indirect N_2O emissions and CH_4 emissions from manure and fermentation per animal per growing period (110 days).

Emission factor	Type	Pig production	Gilt production	Sow production	Tier
Direct N_2O emissions (kg N_2O/animal)	Default	0.0127	0.0162	0.0306	2[(a)]
	Range	0.0064-0.0254	0.0081-0.0323	0.0153-0.0612	
Indirect N_2O emissions (kg N_2O/animal)	Default	0.0159	0.0202	0.0382	2[(a)]
	Range	0.0019-0.0953	0.0024–0.1212	0.0046-0.2294	
CH_4 emissions (kg CH_4/animal)	Default	1.36	0.933	3.66	2
	Range	1.36-5.33	0.933-31.1	3.66-31.1	
CH_4 from fermentation (kg CH_4/animal)	Default	1.5	1.5	1.5	1
	Range	0.75-2.25	0.75-2.25	0.75-2.25	

[a] N excretion in manure was specific for the Netherlands; for emission factors and the gas fractions of volatilisation, default values of the IPCC were used

Emissions due to manure management and enteric fermentation

Handling and storage of manure causes emissions of CH_4 and direct and indirect N_2O emissions (Table 4). Emissions from manure were based on IPCC rules: for CH_4 and N_2O a tier 2 approach was used, whereas for enteric fermentation a tier 1 approach was used. Ranges are caused by epistemic uncertainties around the emission factors (IPCC, 2006). An extended table can be found in the Appendix (Table A.4).

In summary, we identified ranges due to variability in the total feed intake of growing-pigs, yields and N-fertiliser application rates of the five main ingredients of pig diets, and the number of sows and gilts needed per growing-pig (the replacement rate). The mortality rate of the sows and gilts was included in the replacement rate. In addition, ranges due to epistemic uncertainties were found for CO_2 and N_2O emissions of feed-crop production, CH_4 emissions due to enteric fermentation and N_2O, and CH_4 emissions of manure management based on the IPCC tier 1 and tier 2 frameworks. We assumed that all input parameters could vary independently; however, three exceptions were made:

I. In general, when feed intake increases, manure production and N excretion increases as well (CBS, 2010). Therefore, we assumed a proportional relation between feed intake and manure production, i.e. the amount of manure produced (and N excreted) was increased in direct proportion to feed intake, e.g. if feed intake increased 10%, manure production of the growing-pig also increased 10%.

II. In general, when N fertilisation increases, crop yield increases. Therefore, we assumed that the random values drawn for N fertilisation and crop yield followed a similar sampling pattern, e.g. when a high value for N fertilisation was drawn, this resulted in a high value for crop yield and vice versa. This means that if one randomly draws a sample at 2/3 of the uncertainty range for N fertilisation, also for crop yield a sample at 2/3 of the range is selected. But if crop yield is selected first in the trajectory, for example at 1/3 of the uncertainty range, for N fertilisation a sample at 1/3 of the uncertainty range is selected as well.

III. The random values drawn for N fertilisation and crop yield were used to calculate the emissions from cultivation (i.e. CO_2 emission from liming and urea application, direct and indirect N_2O emissions). The emission factors of the CO_2 and N_2O emissions of cropping were still assumed to vary independently from the N fertilisation and the crop yield, because the emission factors also depend on temperature and soil type, etc. Also, the N_2O and CH_4 emission factors from manure management varied independently from the amount of manure, because the emission factors depended not only on the amount of manure but other external factors such as climate conditions (IPCC, 2006).

3 Results and Discussion

3.1 Pork production

GHG emissions per kg body weight (BW) of a growing-pig were 2.61 kg CO_2-eq per kg BW, of which 21% came from crop inputs and 46% came from feed production (Figure 3). Manure management contributed 17% of the total emissions, housing 11%, and enteric fermentation 4.7% (Figure 3). These results corresponded to those found in the literature, in which feed production (crop cultivation, production of crop inputs) and manure management explained most of the emissions (Basset-Mens and Van der Werf 2005; Dalgaard 2007; Van Zanten et al., 2015b).

Based on minimum and maximum values (Tables 1 to 4), minimum and maximum GHG emissions were 1.83 and 5.00 kg CO_2-eq per kg growing-pig, respectively. Estimates for pork production chains are demonstrated to vary from 3.9-10 kg CO_2-eq per kg pork (De Vries and De Boer 2010), converted to kg pork, this resulted in 3.5-9.5 kg CO_2-eq per kg edible product.

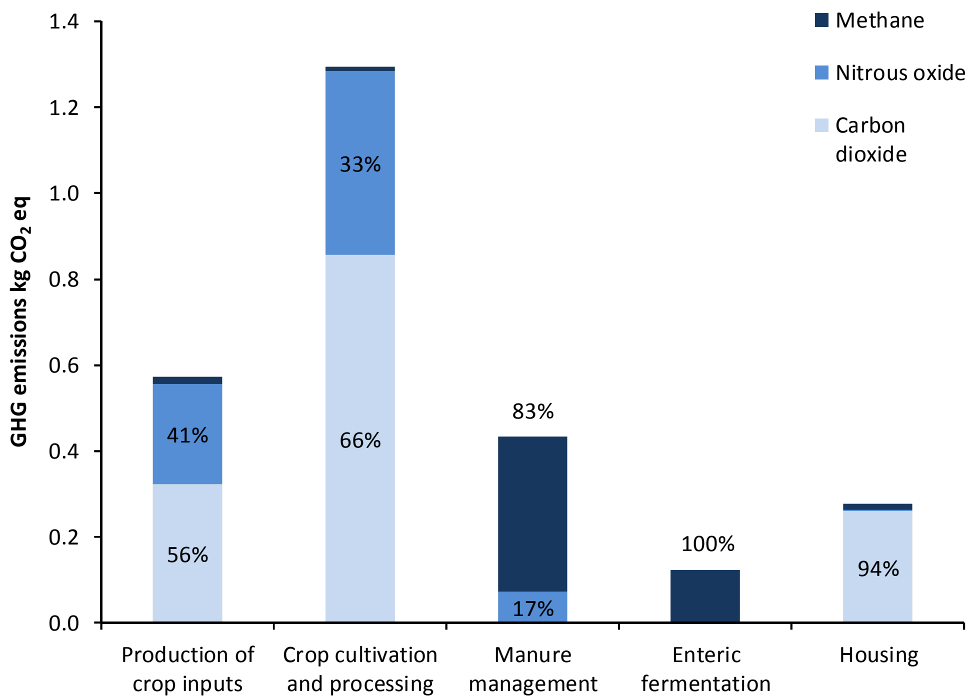

Figure 3. Greenhouse gas emission of 1 kg growing-pig emitted during its growing period (110 days).

3.2 Multiplier method

First we applied MPM, considering the default data only. The most influential parameters were *feed intake (input feed)*, followed by *manure produced by the growing-pig (manure output)* and *yield of maize (maize output)* (respectively #1, #2 and #3, Table 5). Regarding crop inputs, the most influential parameter was the *output of N fertiliser*, which can be interpreted as the efficiency of the N fertiliser production. Regarding crop cultivation, *yield of maize (maize output)*, followed by *yield of barley (barley output)* and *yield of wheat (wheat output)* were most influential. Regarding manure management and fermentation, *manure produced by the growing-pig* was most influential, followed by *CH_4 emissions due to manure* and *enteric fermentation of the growing-pig*.

2

Table 5. Multipliers (η) of the most influential parameters, whose values can be interpreted as follows: increasing an input parameter (i.e. N-fertiliser output) by 1% will change the GWP -0.097%. The ten most influential parameters of the LCA model are shown in bold-italic print. Rank 1 identifies the most influential parameter (i.e. with the largest multiplier). Only parameters with a relatively high influence, i.e. $|\eta| > 0.03$, are shown.

Stage	Process	Flow	Multiplier	Rank within stage	Overall rank	Interpretation
Production of crop inputs	N-fertiliser production	***Fertiliser output***	***-0.097***	***1***	***8***	***Efficiency of N-fertiliser production***
		N_2O emission	+0.061	2		n/a
		CO_2 emission	+0.034	5		n/a
	Diesel production	Diesel output	-0.042	3		Efficiency of diesel production
		CO_2 emission	+0.040	4		n/a
	Urea fertiliser production	Urea fertiliser output	-0.030	6		Efficiency of urea fertiliser production
Crop cultivation and processing	Production of maize	***Maize output***	***-0.14***	***1***	***3***	***Yield of maize***
		Input N-fertiliser	+0.048	5		N-fertiliser rate
		Direct N_2O emission	+0.034	7		n/a
	Production of barley	***Barley output***	***-0.097***	***2***	***9***	***Yield of barley***
	Production of wheat	***Wheat output***	***-0.093***	***3***	***10***	***Yield of wheat***
		Direct N_2O emission	+0.030	8		n/a
	Production of soybean meal	Soybean meal output	-0.053	4		Milling yield of soybean
	Production of soybean	Soybean output	-0.039	6		Yield of soybean (before milling)
	Production of rape meal	Rapeseed meal output	-0.034	9		Milling yield of rapeseed
Production of other feed ingredients	Production of phytase	Phytase output	-0.033	1		n/a
		CO_2 emission	+0.033	2		n/a
Feed production	Pig feed production	***Input maize***	***+0.11***	***1***	***5***	***Maize ratio, pig diet***
		Input wheat	+0.091	2		Wheat ratio, pig diet
		Input barley	+0.062	3		Barley ratio, pig diet
Pork production	Pig production	***Input feed***	***+0.52***	***1***	***1***	***Feed conversion ratio***
		Input piglets	***+0.11***	***2***	***6***	***Number of piglets per pig; piglet mortality***
		Input sows	***+0.10***	***3***	***7***	***Number of sows per pig***[a]
	Sow production	Input feed	+0.081	4		FCR[b], sow
	Piglet production	Input feed	+0.077	5		FCR[b], piglet
Manure management	Manure production, pig	***Manure output***	***+0.15***	***1***	***2***	***Production of manure by***

		CH_4 emission	***+0.12***	***2***	***4***	***growing-pig n/a***
Enteric fermentation	Enteric fermentation, pig	CH_4 emission	+0.043	1		n/a
Housing pig	Energy use	Energy input	+0.064	1		n/a
		CO_2 emission	+0.060	2		n/a

[a] Number of sows required per pig is based on the replacement rate of the sows and on the number of piglets per sow per year

[b] FCR: feed-conversion ratio

3.3 Method of elementary effects

As described (section 2.5.2), defining the ranges for MEE depended on results of MPM (Table 5) and the literature. Based on results of the MPM, we defined ranges for the yields of the five main ingredients of pig diets and N-fertiliser application. Even though indirect N_2O emissions did not show up in the top ten most influential input parameters of the feed ingredients of MPM, we included them as well because Basset-Mens and Van der Werf (2005) showed that the uncertainty ranges of these emissions are high and will influence the results. We defined ranges for the methane emissions of manure and fermentation, and also for the N_2O emission of manure. A range for feed intake of the growing-pig was defined, but not for the maize ratio in the pig diet, because the diets were fixed. We were not able to define ranges for the parameters related to crop inputs, such as N-fertiliser production.

Parameters with the highest elementary effect contributed most to the uncertainty in the results and are considered the most important input parameters: *feed intake of the growing-pig (input feed)* (#1, Figure 4), followed by *methane emissions from manure* (#2, Figure 5), followed by *N-fertiliser input for maize cultivation* (#3, Figure 6).

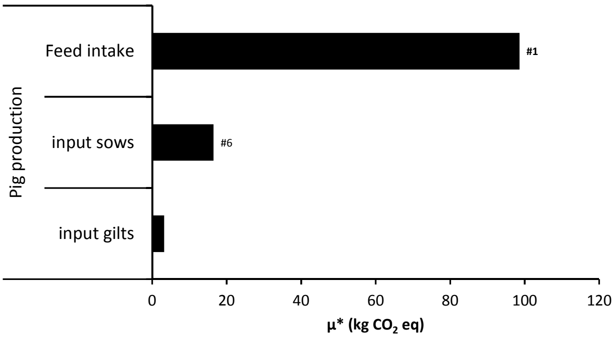

Figure 4. Elementary effects (μ*) of technical parameters. Feed intake is most important parameter in the LCA model. Numbers at the end of bars indicate the overall rank for the 10 parameters that contributed most to output uncertainty.

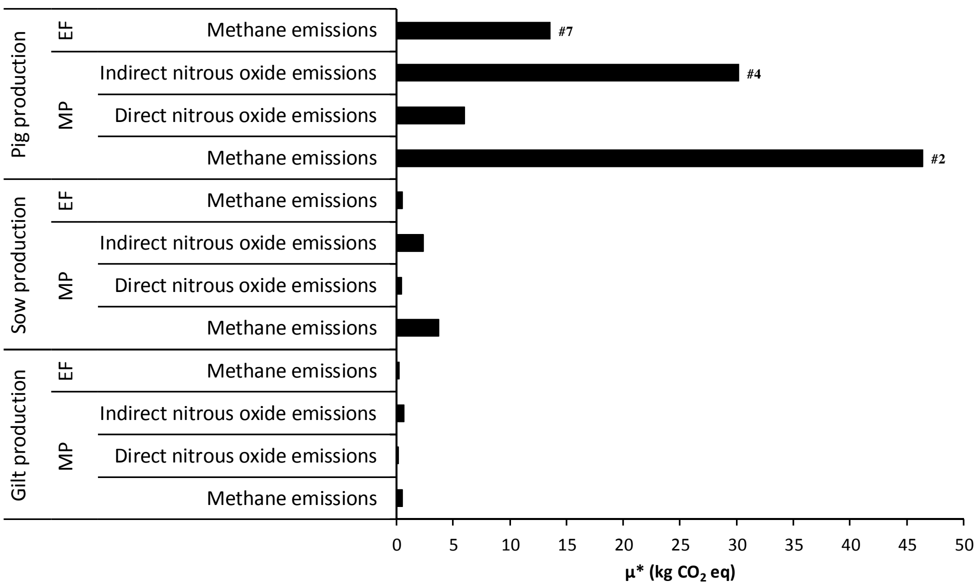

Figure 5. Elementary effects (μ*) of parameters causing uncertainty in the results of manure management and enteric fermentation. MP: manure production; EF: enteric fermentation. Numbers at the end of bars indicate the overall rank for the 10 parameters that contributed most to output uncertainty.

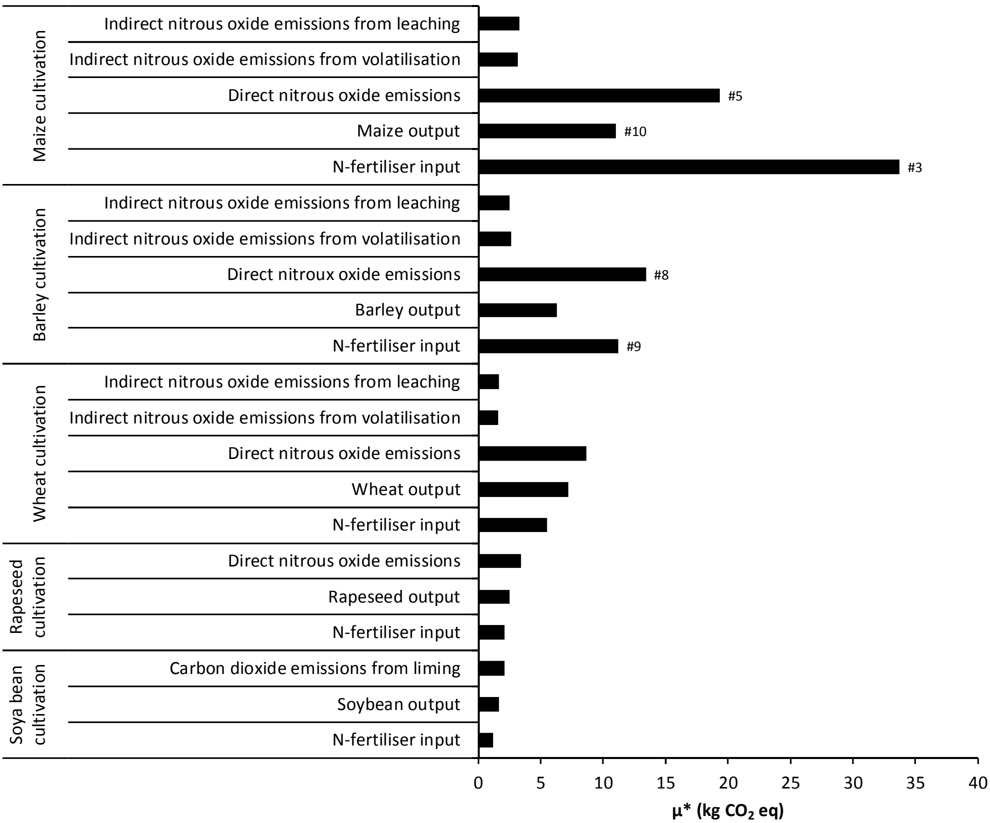

Figure 6. Elementary effects (μ*) of parameters causing most uncertainty (i.e. μ*>1) in the result of crop cultivation. Numbers at the end of bars indicate the overall rank for the 10 parameters that contributed most to output uncertainty.

3.4 Discussion of the sensitivity-analysis methods

Applying MPM for sensitivity analysis overcomes arbitrary choices of selecting a subset of input parameters, as done in traditional OAT sensitivity-analysis methods. One disadvantage of MPM is that the effect of uncertainty ranges around input parameters on model output is not included. MEE allowed us to include the uncertainty ranges of input parameters that were available in the single-issue LCA model. However, one disadvantage of MEE is that it is based only on minimum and maximum values, thus excluding a distribution function or an average value. Other methods for sensitivity analysis are available which belong to the area of global sensitivity analysis, such as squared standardized regression coefficients (Saltelli et al., 2008), which quantify how much each input parameter contributes to output variance (Campolongo et al., 2007; Saltelli et al., 2008). To apply a global sensitivity analysis, more data are required, such as the standard deviation and a distribution function. Because these types of data were not available in this study, we could apply this method. However, we were interested mainly in influential input parameters that could give direction for future innovations; important parameters that could improve farm management strategies and improve reliability of results, which could also be derived with MEE.

There are two disadvantages to the way in which we applied MEE. First, we were not able to identify ranges for all input parameters; therefore, we might have missed potentially important input parameters. Second, we assumed that all input parameters either varied independently or were directly related (i.e. N fertiliser and crop yield), which probably overestimates what happens in reality. However, MEE is less suitable for implementing correlations than more data-intensive global sensitivity-analysis methods, such as using the squared standardized regression coefficients as a proxy for a sensitivity index.

One of the most influential input parameters was the amount of manure produced by the growing-pig, which was directly related to feed intake. That increased feed intake results in increased manure production and N excretion is plausible; however, assuming that it does so in direct proportion remains questionable. In addition to feed intake, factors such as water intake can also change the amount and N content of manure. However, ignoring a relation between feed intake and manure production and N excretion would have resulted in underestimating CH_4 and indirect and direct N_2O emissions. To what extent manure production and N excretion are over- or underestimated in relation to feed intake, however, remains unclear.

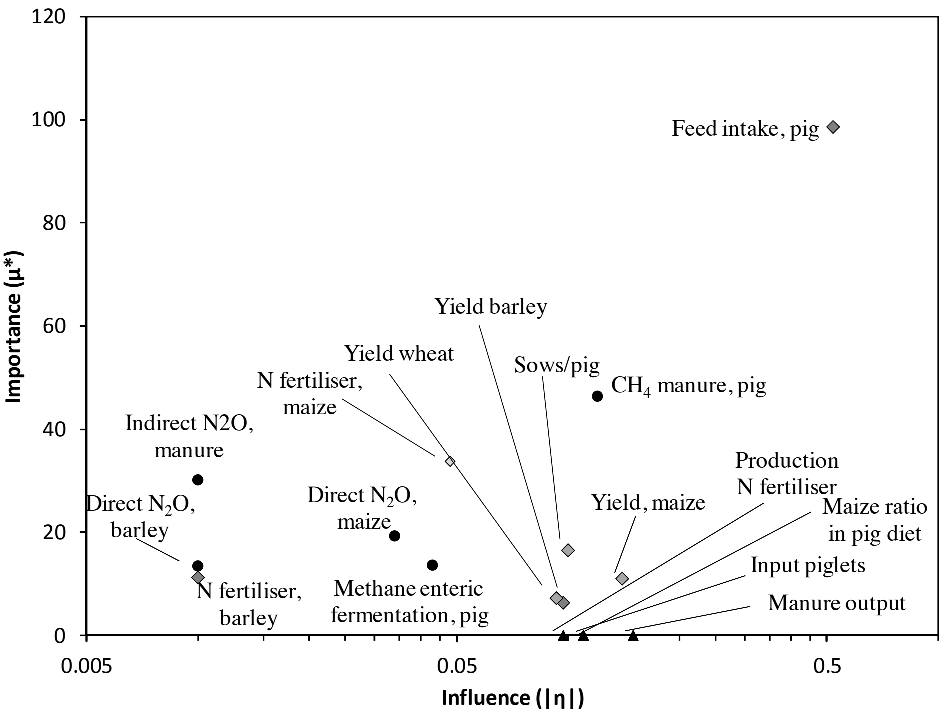

Figure 7. Most essential parameters in the LCA model of pork production. Parameters along the horizontal axis (log-scale) are the most influential parameters identified by MPM. Parameters along the vertical axis are the most important parameters identified by MEE. *Circles*: parameters containing epistemic uncertainty; *diamonds*: parameters containing variability. Parameters on the horizontal axis (*triangles*) are those for which no ranges could be defined. MPM: multiplier method; MEE: method of elementary effects.

3.5 Combining MPM and MEE

The results of MPM identified a different set of parameters than MEE. By combining results of the two sensitivity-analysis methods, we could extract the most essential parameters to identify GHG mitigation strategies for pork production and improve reliability of the results. The most influential parameters are *feed intake of the growing-pig*, followed by *manure produced by the growing-pig* and *yield of maize* (Figure 7), while the most important parameters are *feed intake of the growing-pig*, followed by *methane emissions of manure* and by *N fertiliser of maize*. *Feed intake of the growing-pig* is therefore considered the most essential parameter in the LCA model of GHG emissions of pork production. A change in feed intake immediately affects the amount of feed produced and the corresponding emissions of the feed ingredients.

Parameters with input uncertainties that affect the output (i.e. high importance), such as the direct N_2O emissions of barley and maize (from leaching and volatilization), have in fact a low influence. Applying only MPM would have led to overlooking these parameters. In contrast, parameters that have relatively high influence, but for which no uncertainty ranges could be defined (e.g. N-fertiliser production), might have been underestimated if only MEE had been applied.

3.6 Formulating mitigation options and improving reliability

The most influential parameters (Figure 7, horizontal axis) have the most impact when they are reduced. Innovation options to improve the most influential parameters, such as decreasing feed intake of growing-pigs, decreasing the amount of manure produced, increasing yields of feed ingredients in pig diets, decreasing the number of sows needed per growing-pig, decreasing piglet mortality, and increasing efficiency of N-fertiliser production, will have a large effect on results. These mitigation strategies result in increased efficiency that will have an effect throughout the production chain.

Mitigation options via e.g. management strategies can be formulated by looking at the most important parameters (Figure 7, vertical axis) affected by natural variability (Figure 7, *diamonds*), such as feed intake and fertiliser application. Natural variability in parameters can be caused by variability in climate, soil types, or temperature, or differences in genetics, geography or farm management. Farmers can strive to reduce environmental impacts of the pig production chain by adapting their management strategies towards those of the most successful farmers. For example, farmers can improve the feed conversion ratio (kg dry matter feed intake per kg growth), but doing so is not easy, because it depends on several factors, such as feed quality, feed access, pig health, and climate conditions e.g. temperature, and humidity of the stalls.

Reliability of GHG assessment can be improved by looking at parameters with epistemic uncertainties (Figure 7, *circles*). Epistemic uncertainties around emission factors for indirect N_2O emissions and CH_4 emissions of manure, and direct N_2O emissions of maize and barley cultivation have the most effect on the reliability of results. Decreasing uncertainty ranges of these emission factors would decrease those around the output, hence provide more reliable conclusions. However, reducing uncertainty around the indirect N_2O emission of maize production would not mean that GHG emissions are actually reduced, only that the results are more reliable. Estimates can be improved by taking measurements in the field. Reliability could also be improved by improving data quality of the most essential parameters.

Mitigation strategies, either in the form of technical innovations or improved management practices related to feed intake, would reduce GHG emissions the most. Feed cultivation has higher impacts than other stages of pork production. However, our results showed that besides feeding strategy, mitigation strategies related manure management are also important. If feed-related mitigation strategies are assessed, possible trade-offs with manure management should also be considered, as they might have a high impact on the results.

4 Conclusion

We applied two methods for sensitivity analysis, MPM and MEE. Combining both methods allowed us to determine the most essential parameters in the model, from which we could derive mitigation options based on innovation and management strategies, and to formulate options for improving reliability of estimates of GHG emissions of a pork production chain. Mitigation options based on innovation (e.g. novel feeding strategies) would be suggested for the most influential parameters in the model (identified by MPM): feed intake, amount of manure produced by growing-pigs, crop yields of the main feed ingredients, number of sows required for one growing-pig and fertiliser-production efficiency. Mitigation options based on management strategies (e.g. reducing mortality rate) would be suggested for technical parameters with high variability (identified by MEE), such as feed intake, crop yields and number of sows per pig. The uncertainty ranges can be used as margins of improvement within the pork production chain. In addition to that, MEE showed that reliability could be improved most when uncertainty ranges around direct and indirect N_2O emissions of the main feed crops in the pig diet and the CH_4 emissions of manure production are reduced. Combining two sensitivity-analysis methods identified the most essential input parameters in the pork production chain, while allowing for uncertainties around input data. With this combined analysis, potential targets for mitigation options via innovations and management strategies were derived, and parameters were identified that improved reliability of the results.

Appendix

Data required to determine technical parameters

Table A.1. Average diet composition pigs, gilts, sows and piglet; the five main ingredients are given in bold. Reference: Van Zanten et al. (2015b) and for growing-pigs (Bikker et al., 2011).

Ingredient	Growing-pig (%)	Gilt (%)	Sow (%)	Piglet (%)
Animal fat	2.00	2.00	0.70	2.00
Barley	**12.9**	**6.78**	**13.7**	**32.1**
Bread meal	2.75	2.25	1.58	1.75
Maize	**26.7**	**25.0**	**21.1**	**21.5**
DL-Methionine	0.03	0.02	0.01	0.14
Lactic Acid	-	-	-	1.00
Limestone	1.06	0.90	1.11	1.00
L-Lysine HCL	0.34	0.30	0.24	0.45
L-Threonine	0.06	0.05	0.07	0.12
L-Tryptofaan	0.02	-	-	-
Monocalciumphosphate	0.13	0.11	0.38	0.66
Palm kernel expeller	1.00	-	3.25	-
Phytase	0.62	0.65	0.65	0.65
Potato protein	-	-	-	1.35
Premix	0.40	0.40	0.40	0.40
Rapeseed expeller	1.00	-	0.84	-
Rapeseed meal	**10.2**	**10.0**	**1.30**	**-**
Salt	0.32	0.30	0.54	0.60
Soybean hulls	-	2.54	6.00	-
Soybean meal	**7.50**	**4.25**	**3.70**	**12.9**
Soybeans heat treated	-	-	-	0.11
Sugar beet pulp	-	5.00	8.82	1.00
Sugarcane molasses	2.90	2.00	2.40	1.44
Triticale	1.50	1.13	-	-
Wheat	**20.9**	**20.4**	**12.4**	**11.0**
Wheat middling	3.69	10.2	15.7	5.00
Sunflower oil	0.69	0.41	0.49	0.50
Sunflower seed meal	3.29	5.00	3.49	3.00
Whey powder	-	-	-	1.00

Table A.2. Technical parameters for crop production. Sum of transportation of crop ingredients (N, P, K, lime, pesticides) to the farm, transportation of to the feed factory or drying (barley, maize and wheat) or to feed mill (rapeseed, soy), and from the factory to the Netherlands.

	Barley	Maize[a]	Rapeseed	Soy	Wheat	Ref.
Land of origin	France	France	France	Brazil[b]	France	GL[c]
cr_{yield} (kg dm/ha · year)	5520	6518	3040	4800	6010	GL[c]
Allocation (%)	-	-	0.25	0.59	-	CvM[d]
Seeds (kg/ha·year)	125	20	3	110	140	GL[c]
Pesticides (kg act. sub./ha·year)	8.96	1	1.13	5	1.97	GL[c]
N fertilizer (kg/ha·year)	130	150	73.4	9	55	GL[c]
Urea (kg N/ha·year)	24.7	0	91.6	18	110	CvM[d]
P fertilizer (kg/ha·year)	37	56.8	50	180	26	GL[c]
K fertilizer kg/ha·year	34	63	50	180	24	GL[c]
Lime (kg/ha·year)	298	298	298	2160	298	CvM[d]
N manure (kg/ha·year)	10	10[a]	16	0	10	GL[c]
Diesel (kg/ha·year)	84	85	92	160	83	GL[c]
Agricultural machinery (kg/ha·year)	18.7	21.7	20.4	39	18.6	GL[c]
Yield before drying (kg dm/ha·year)	-	7621	-	-	-	CvM[d]
Electricity for drying (kWh/kg)	-	0.008	-	-	-	CvM[d]
Lorry (tkm)	785	1130	420	529	785	CvM[d]
Rail (tkm)	8416	10948	3661	0	8808	CvM[d]
Sea (tkm)	0	0	0	45492	0	TS[e]

[a] Manure for corn: assumption similar to wheat/barley
[b] Central-West Brazil
[c] Garcia-Launay et al. (2014)
[d] Van Middelaar et al. (2013)
[e] TS: assumption in this study

Table A.3. Default values and ranges for breeding and housing (Van Zanten et al., 2015b).

	Value	Ranges	Unit
Breeding[a]	0.01731	0.01454 - 0.02363	Number of gilts/pig
	0.03397	0.02853 - 0.04637	Number of sows/pig
	1.022	-	Number of piglets/pig
Housing[b]	0.8	-	m^2/pig
	2.25	-	m^2/gilt
	2.25	-	m^2/sow
	0.35	-	m^2/piglet

[a] The amount of sows and gilts required for the production of one pig included death rate
[b] For piglets, gilts and sows we compensated for the difference in m^2 used per animal place in comparison with the m^2 used per growing-pig place based on Dutch regulations (policy document, 2007)

IPCC Tier 1 and Tier 2 equations to determine the emission factors

Crop production

The direct N_2O emissions from crop production are quantified using equation 11.1 (IPCC, 2006) (adjusted amount of N in mineral soils that is mineralised is zero: $F_{som} = 0$):

$$\boxed{N_2O_{direct,crop\ production} = \frac{44}{28}(N_{sf} + N_m + N_{cr})EF_1} \quad \text{(A.1)}$$

where: N_{sf} (kg N/year) is the amount of N in synthetic fertilizer, N_m (kg N/year) the amount of N in manure, N_{cr} (kg N/year) the amount of N in crop residues and EF_1 (kg $N_2O -$ N/(kg N year)) the emission factor of direct N from fertilizer, manure and crop residues. The amount of N in crops (N_{cr}) is estimated by equation 11.6 (adjusted: no area burnt $area_{burnt} = 0$, pastures are renewed every year: $frac_{renew} = 1$, considering 1 hectare: $area = 1$):

$$N_{cr} = cr_{yield}\left[R_{ag} \cdot N_{ag}(1 - frac_{rem}) + R_{bg} \cdot N_{bg}\right] \quad \text{(A.2)}$$

where: cr_{yield} (kg dm/ha) is the crop yield, R_{ag} (kg dm/dm) the ratio of above ground residue dry matter to the harvest yield, N_{ag} (kg N/kg dm) is the N content of above-ground residues, $frac_{rem}$ (%) is the fraction of above-ground residues that is removed, R_{bg} (kg dm/kg dm) is the ratio of below-ground crop residues to harvested yield and N_{bg} (kg N/kg dm) the N content of below-ground crop-residues. R_{ag} is calculated by:

$$R_{ag} = \frac{AG_{DM} \cdot 1000}{cr_{yield}} \quad \text{(A.3)}$$

where AG_{DM} (Mg/ha) is the above-ground residue, which can be estimated by:

$$AG_{DM} = \frac{cr_{yield} \cdot slope}{1000} + intercept \quad \text{(A.4)}$$

and R_{bg} can be estimated by:

$$R_{bg} = R_{BG-BIO} \cdot \frac{AG_{DM} \cdot 1000 + cr_{yield}}{cr_{yield}} \quad \text{(A.5)}$$

where R_{BG-BIO} (%) is the ratio of belowground residues to aboveground biomass.

The indirect N_2O emissions from crop production come from leaching and volatilization. Leaching is quantified using equation 11.10 (adjusted, leaching of mineralised N and N leaching from urine and dung is zero: $F_{som} = 0$ and $F_{PRP} = 0$):

$$\boxed{N_2O_{leaching,crop\ production} = \frac{44}{28}\left(N_{sf} + N_m + N_{cr}\right) \cdot Frac_{leach} \cdot EF_5} \quad (A.6)$$

where $Frac_{leach}(\%)$ is the fraction of N added to managed soils and EF_5(kg N_2O/kg N leached) is the emission factor for N_2O emissions from leaching. The indirect N_2O emissions due to volatilization are calculated with equation 11.9 (adjusted, $F_{PRP} = 0$):

$$\boxed{N_2O_{volatilization,crop\ production} = \frac{44}{28}\left(N_{sf} \cdot Frac_{gasf} + N_m + Frac_{gam}\right) \cdot EF_4} \quad (A.7)$$

where $\mathrm{Frac_{gasf}}(\%)$ is the fraction of synthetic fertilizer that volatilizes, $\mathrm{Frac_{gam}}(\%)$ is the fraction of manure that volatiles and $EF_4(\mathrm{kg\ N - N_2O/(kg\ NH_3 - N\ +\ NO_x - N\ volatilised)})$ is the emission factor of N_2O emissions of atmospheric deposition of N.

The CO_2 emission factor for liming are calculated using equation (11.12) (adjusted, no dolomite liming $M_{dolomite} = 0$):

$$\boxed{CO_{2\,liming} = 1000 \cdot \frac{44}{12} \cdot M_{lime} \cdot EF_{lime}} \quad (A.8)$$

where M_{lime}(ton C/year) is the annual amount of limestone ($CaCO_3$) and $\mathrm{EF_{lime}}$(ton C/ton limestone) is the emission factor. The annual CO_2 emissions due to urea fertilization are given by:

$$\boxed{CO_{2\,urea\ fertilization} = 1000 \cdot \frac{44}{12} \cdot M_{urea} \cdot EF_{urea}} \quad (A.9)$$

where M_{urea}(ton C/year) is the amount of urea applied per year and EF_{urea}(ton C/ton urea) is the emission factor.

Manure management

Direct N_2O emissions from manure management (equation 10.25, adjusted) (IPCC, 2006):

$$N_2O_{direct,manure} = \frac{44}{28}(N_{ex})EF_3 \quad (A.10)$$

where N_{ex} is the amount of excreted N in manure per animal per year and EF_3 is the emission factor for direct N_2O emissions from manure (kg N_2O/kg).

Indirect N_2O emissions from manure management (equation 10.26 & 10.27, adjusted)

$$N_2O_{indirect,manure,volatilization} = \frac{44}{28} N_{ex} \cdot Frac_{gam} \cdot EF_4 \quad \text{(A.11)}$$

where $Frac_{gam}$ is the fraction of manure that volatilizes. The CH_4 emission for manure management was available for Tier 2 (equation 10.23, adjusted):

$$CH_{4_{manure}} = V_s \cdot 365 \cdot B_o \cdot 0.67 \cdot \frac{MCF}{100} \quad \text{(A.12)}$$

where V_s is the daily volatile solid excreted (kg DM/day), B_o maximum methane producing capacity for manure (m^3 CH_4/kg), MCF the methane conversion factor (%). Results can be found in Table A.4.

Table A.4. Direct/indirect N_2O emissions and CH_4 emissions (IPCC Tier 2) from manure for growing-pigs, gilts, and sows (kg per year).

	Pig	**Gilt**	**Sow**	**Reference**
Manure (kg)	356	420	1649	RIVM[a]
N-content (%)	0.0114	0.0122	0.0059	RIVM[a]
N-content (kg)	4.04	5.14	9.73	
EF_3(kg N_2O/kg N)	0.002	0.002	0.002	Table 10.21[b]
	(0.001 - 0.004)	(0.001 - 0.004)	(0.001 - 0.004)	
Direct N_2O (kg N_2O/year)	**0.0127**	**0.0162**	**0.0306**	
	(0.0064 - 0.0254)	**(0.0081 -0.0323)**	**(0.0153 - 0.0612)**	
EF_4 (kg N_2O-N/kg N)	0.01	0.01	0.01	Table 11.3[b]
	(0.01 - 0.05)	(0.01 - 0.05)	(0.01 - 0.05)	
$Frac_{gam}$(%)	0.25	0.25	0.25	Table 10.22[b]
	(0.15 - 0.3)	(0.15 - 0.3)	(0.15 - 0.3)	
Indirect N_2O (kg N_2O/year)	0.0159	0.0202	0.0382	
	(0.0019 - 0.0953)	(0.0024 - 0.1212)	(0.0046 - 0.2294)	
V_s	15.29	10.51	41.22	
B_o($m^3 CH_4$/kg manure)	0.34	0.34	0.34	
MCF	0.39	0.39	0.39	
CH_4 (kg CH_4/year)	**1.36**	**0.933**	**3.66**	
	(1.36 - 5.33)	**(0.933 - 31.1)**	**(3.66 - 31.1)**	

[a] Coenen et al. (2013)
[b] IPCC (2006)

Chapter 3

Environmental impact of replacing soybean meal with rapeseed meal in diets of finishing-pigs

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Abstract

The major impact of the livestock sector on the environment may be reduced by feeding agricultural co-products to animals. Since the last decade, co-products from bio-diesel production, such as rapeseed meal (RSM) became increasingly available in Europe. Consequently, an increase in RSM content in livestock diets was observed at the expense of soybean meal (SBM) content. Cultivation of SBM is associated with high environmental impacts, especially when emissions related to land use change (LUC) are included. This study aims to assess the environmental impact of replacing SBM with RSM in finishing-pig diets. As RSM has a lower nutritional value, we assessed the environmental impact of replacing SBM with RSM using scenarios that differed in handling changes in nutritional level. Scenario 1 (S1) was the basic scenario containing SBM. In scenario 2 (S2), RSM replaced SBM based on crude protein content, resulting in reduced energy and amino acid content and hence an increased feed intake to realize the same growth rate. The diet of scenario 3 (S3) was identical to S2, however we assumed that pigs were not able to increase their feed intake leading to reduced growth performance. In scenario 4 (S4) the energy and amino acid content were increased to the same level of S1. Pig performance were simulated using a growth model. We analysed the environmental impact of each scenario using life cycle assessment, including processes of feed production, manure management, piglet production, enteric fermentation, and housing. Results show that, expressed per kg of body weight, replacing SBM with RSM in finishing-pig diets marginally decreased global warming potential (GWP) and energy use (EU) but decreased land use (LU) up to 12%. Between scenarios, S3 had most potential to reduce the environmental impact, due to a lower impact per kg of feed and an increased body protein to lipid ratio of the pigs resulting in a better feed conversion ratio. Optimisation of the body protein to lipid ratio, therefore, might result in an reduced environmental impact of pig production. Furthermore, the impact of replacing SBM with RSM changed only marginally when emissions related to direct (up to 2.9%) and indirect LUC (up to 2.5%) were included. In case we evaluated environmental impacts of feed production only, which implies excluding other processes along the chain as is generally found in literature, GWP decreased up to 10% including LUC, EU up to 5%, and LU up to 16%.

Implications

Livestock production has a major impact on the environment which can be reduced by feeding co-products. Rapeseed meal (a co-product from bio-diesel production) increasingly replaces soybean meal in pig feed. This may reduce the environmental impact of pig production. Results of this study show that replacing soybean meal with rapeseed meal reduces land use up to 12%. However, it only marginally decreases global warming potential (up to 1-3%, depending whether or not emissions related to LUC are included) and energy use (up to 2%).

1 Introduction

Livestock production causes severe environmental pressure via emissions to air, water, and soil (Steinfeld et al., 2006). The livestock sector is responsible for about 15% of the total anthropogenic emissions of greenhouse gases (Gerber et al., 2013), which are mostly related to production and utilization of feed (De Vries and De Boer, 2010). The livestock sector also increasingly competes for scarce resources such as land, water, and fossil-energy (Steinfeld et al., 2006; De Vries and De Boer, 2010). The challenge is to reduce emissions and to increase efficient use of resources.

Feeding co-products from arable production or the human food processing industry to livestock may lower the environmental impact (Elferink et al., 2008). Co-products from bio-diesel production, such as rapeseed meal (RSM) became increasingly available during the last decade in Europe (Makkar et al., 2012). Consequently, the RSM content of livestock diets increased (Vellinga et al., 2009). In 1994, the RSM content of livestock diets in the Netherlands was 2% for dairy cows, 5% for poultry and pigs, whereas in 2007 it increased to 3% for dairy cows, 7% for poultry and 12% for pigs (Vellinga et al., 2009). RSM is a protein-rich feed ingredient and will replace other protein-rich ingredients, such as soybean meal (SBM) (Thamsiriroj and Murphy, 2010; Reinhard and Zah, 2011). Cultivation of SBM has a high environmental impact partly due to large transport distances, its high economic value when based on economic allocation (Cederberg and Flysjo, 2004; Van der Werf et al., 2005; Vellinga et al., 2009), and due to emissions related to land use change (LUC) such as deforestation in South America (Foley et al., 2007; Prudêncio da Silva et al., 2010). Due to its high environmental impact, it is expected that replacing SBM with RSM will lead to a decrease in environmental impact.

This study, therefore, aims to assess the environmental impact of replacing SBM with RSM in finishing-pig diets in Europe. We focused especially on finishing-pigs as they use about 60% of the total feed in the pig production chain. Because RSM has lower nutritional values than SBM, i.e. lower crude protein and essential amino acid contents, and a lower net energy value, replacing SBM with RSM changes the nutritional value of the diet and/or affects feed intake and growth performance of the finishing-pig. Scenarios with different diet compositions and nutritional levels were used to assess the environmental impact of replacing SBM with RSM. This study focused on pigs, as for this species no studies about the substitution of SBM with RSM are available, while the largest increase in use of RSM occurred here. Life cycle assessment (LCA) was performed for all four scenarios, regarding greenhouse gas (GHG) emissions, land use (LU), and energy use (EU).

2 Material and methods

2.1 Scenario definition

Four scenarios were developed, a reference scenario (S1) with SBM and three alternative scenarios (S2, S3, and S4) in which SBM was replaced with RSM (Figure 1). S1 was based on Dutch average standards of diets for finishing-pigs, and contained 15% SBM, 9.50 MJ net energy (NE), and 7.59 g standard ileal digestible lysine (SID LYS) per kg of feed while pigs were fed ad libitum (Vellinga et al., 2009; CVB, 2010; Peet-Schwering et al., 2012). Definition of S2, S3, and S4 contained three steps. First, we determined how much RSM is needed to replace SBM (identical for S2, S3, and S4). Second, we described routes chosen in S2, S3, and S4 to handle differences in nutritional levels of diets resulting from the difference in nutritional value between SBM and RSM. Third, the final diet was formulated using an optimization method, taking into account constrains formulated during the first and second step.

Step one

15% SBM and 8% barley were replaced with 23% RSM based on their crude protein (CP) content. The replacement rate was obtained as follow. S1 contains 15% SBM, which equals 70 g CP using a CP content of SBM of 464 g/kg (CVB, 2010). To replace 70 g of CP from SBM, we need 208 g RSM with a CP content of 335 g/kg (CVB, 2010). In short, 150 g of SBM was replaced with 208 g of RSM per kg feed, implying a reduction of 58 g of other feed ingredients and their associated CP content. We assumed this to be 58 g of barley, with a CP content of 104 g/kg. This again results in a loss of CP from barley, and therefore, fine-tuning this exchange can continue eternally. Finally, therefore, the reference diet should contain a minimum of 15% SBM and 8% barley (70 g CP from SBM and 8 g CP from barley) which was assumed to be replaced with a minimum of 23% (77 g CP) of RSM in the diets of the three scenarios.

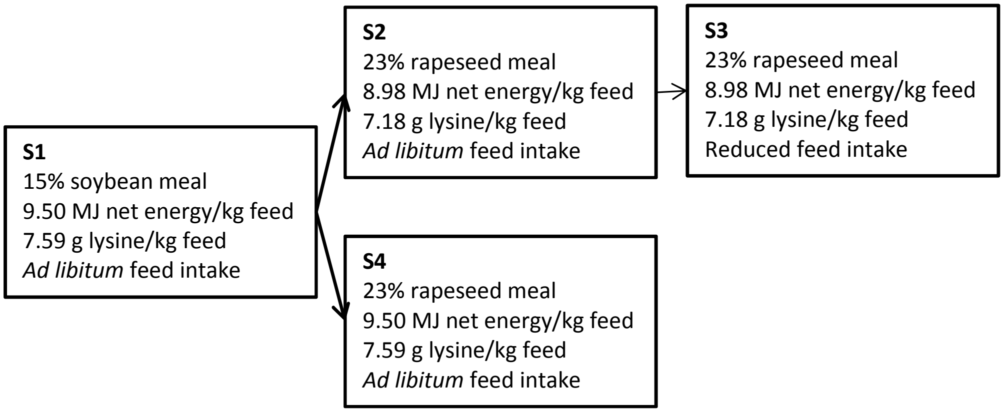

Figure 1. Four scenarios of replacing soybean meal with rapeseed meal in diets of finishing-pigs**.** Scenario 1 (S1) contains soybean meal (SBM) which is replaced with rapeseed meal (RSM) in scenarios S2, S3 and S4. In S2 the same daily net energy intake as in S1 is realised by an increased feed intake. In S3 the same daily feed intake as in S1 and lower net energy intake is realised. In S4 the diet is optimised to contain the same net energy and lysine content as in S1.

Step two

Handling changes in nutritional levels of diets (Figure 1). Due to the differences in nutritional value between SBM and RSM, replacing SBM with RSM based on crude protein affects the NE content and amino acid content of the diet. Losses in NE can be compensated by adding fat, whereas losses in amino acids can be compensated by adding industrial amino acids (AA) as is usually done in practice. Mosnier et al. (2011) and Meul et al. (2012), however, found a high carbon footprint of synthetic amino acids (SAA) due to the energy intensive production process. On the other hand, Garcia-Launay et al. (2014) concluded that using SAA reduced the carbon footprint of pig production. Therefore, we have chosen different routes to handle the difference in nutritional level of the diet. In S2, we did not compensate for the loss in nutrient density of the diet. Therefore, the nutritional value per kg feed was reduced to 8.98 MJ NE and 7.18 g SID LYS, and therefore, an increased feed intake was required to realize the same growth performance. However, if a diet contains less than approximately 9 MJ NE per kg feed, pigs might not be able to increase their feed intake resulting in a decreased NE intake per day (Quiniou and Noblet, 2012). In S3 (identical to S2) we, therefore, assumed that pigs were not able to increase their feed intake leading to reduced growth performance. In scenario 4 (S4) the energy and amino acid contents were increased to the same level of S1. In each scenario the amount of SID LYS was related to NE, using a minimum of 0.8 g SID LYS per MJ of NE (CVB, 2010).

Step three

Final diet composition. Diets in S1, S2, S3, and S4 were formulated using a commercial linear programming tool (i.e. Bestmix®, Adifo, Maldegem, Belgium), which optimizes a diet by minimizing the cost price of the diet (Table 1). In the Appendix the precise nutritional value of each diet is described (Table A.1). The price of ingredients was based on the average of a quarterly published pricelist of 2012 (Nuscience, 2012). Diets had to meet requirements for SID methionine and cystine 62%, threonine 65%, and tryptophan 20% relative to SID lysine (CVB, 2010). Furthermore, dietary restrictions were applied based on regular Dutch practice in finishing-pig production: a diet could contain maximally 30% maize, 40% wheat, 40% barley, 10% peas, 2% molasses, 500 FTU phytase per kg, and should contain 0.4% premix to provide minerals and vitamins.

3

Table 1. Diet composition of the scenarios in which soybean meal is replaced with rapeseed meal at equal dietary protein content.

Ingredients, %	Scenario 1	Scenario 2	Scenario 3	Scenario 4
Rapeseed meal, 34% CP	-	23.00	23.00	23.00
Soybean meal, 46% CP	15.00	-	-	-
Peas	9.36	10.00	10.00	10.00
Maize	30.00	30.00	30.00	30.00
Wheat	29.74	30.43	30.43	30.24
Wheat middlings	0.90	2.23	2.23	
Barley	10.10	-	-	-
Sugarcane molasses	2.00	2.00	2.00	2.00
Phytase premix	0.65	0.53	0.53	0.65
Vitamins and minerals premix	0.40	0.40	0.40	0.40
Animal fat	-	-	-	2.09
Limestone	1.24	0.88	0.88	0.96
Salt	0.37	0.35	0.35	0.38
Monocalcium phosphate	0.11	-	-	0.01
L-Lysine HCL	0.10	0.16	0.16	0.22
L-Threonine	-	-	-	0.02
DL-Methionine	0.03	-	-	0.01

2.2 Growth performance

To analyse the impact of each scenario on growth performance of pigs, the model 'INRAporc' (Van Milgen et al., 2008) was used. This model simulates how nutrients are used for protein deposition (PD) and lipid deposition (LD), and for supporting other functions (i.e. maintenance, physical activities, and protein deposition costs). Potential PD, energy supply (NE intake), and amino acid supply are driving forces that determine the rate of PD and LD. Potential PD is defined as the PD when the animal is capable of expressing its full growth potential under ad libitum feeding. To define the parameters used in INRAporc, we characterized finishing-pigs based on data of Peet-Schwering et al. (2012). Pig characterization in INRAporc was best represented by means of late maturing gilts. The following input parameters were used: age at start 70 days, weight at start 23.6 kg, final age 180 days, precocity of 0.0135 per day, and a mean PD of 122 g per day. Feed intake was calculated as $Y=aX^b$, with factor a is 2.428 and factor b is 0.497. Factor a and b were based on a feed intake of 17 MJ NE at 50 kg and 24 MJ NE at 100 kg (Peet-Schwering et al., 2012). Until gilts reached a weight of 50 kg body weight, starter feed was used, including 9.68 MJ NE and 9.48 SID LYS. Above 50 kg body weight, the four scenarios were implemented. Feed intake and growth performance per scenario are presented in Table 2.

Table 2. Impact of replacement of soybean meal (scenario 1) with rapeseed meal (scenario 2, 3, and 4) on growth performance of gilts from 24 kg body weight simulated with INRAporc.

	Scenario 1	Scenario 2[a]	Scenario 3[b]	Scenario 4[c]
Diet NE content, MJ/kg	9.50	8.98	8.98	9.50
SID LYS g/kg	7.59	7.18	7.18	7.59
Total feed intake (kg)	226	237	226	226
Body gain (g/d)	840	840	820	840
Feed conversion ratio	2.44	2.55	2.49	2.44
Final body mass (kg)	116.4	116.4	114.3	116.4
Protein mass (kg)	19.17	19.17	19.05	19.17
Lipid mass (kg)	19.13	19.13	17.03	19.13

[a] The same daily net energy intake as in S1 is realised by an increased feed intake, resulting in a similar growth performance

[b] The same daily feed intake as in S1 and lower net energy intake is realised, resulting in a decreased growth performance

[c] The diet is formulated to contain the same net energy and lysine content as in S1, resulting in a similar feed intake and growth performance

2.3 Assessing environmental impact of dietary scenarios

To assess the environmental impact for each scenario, an attributional LCA was used. An attributional LCA is an internationally standardized holistic method to evaluate the environmental impact during the entire production chain (Guinée et al., 2002; Bauman and Tillman, 2004). During the life cycle of a product two types of environmental impacts are considered: use of resources such as land or fossil fuels, and emissions of pollutants (Guinée et al., 2002). We assessed GHG emissions, energy use (EU), and land use (LU). Emission of GHGs, EU, and LU were chosen as examples as the livestock sector contributes significantly to both climate change and LU worldwide (Steinfeld et al., 2006). Furthermore, EU was used as it influences GWP considerably. The following GHGs were included: carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O). These GHGs were summed up based on their equivalence weighting factors in terms of CO_2-eq (100 years' time horizon): i.e. 1 for CO_2, 25 for CH_4, and 298 for N_2O (Forster et al., 2007). LU was expressed in m^2.year / kg body weight and EU was expressed in MJ per kg of body weight. Besides expressing the environmental impact per kg of body weight we assessed the impact per kg of protein as livestock products contribute especially to the protein demand of humans (De Vries and De Boer, 2010). In case of a multifunctional process (e.g. production of soybean oil and meal), economic allocation was used, which is the partitioning of environmental impacts between co-products based on the relative economic value of the outputs (Guinée et al., 2002). Economic allocation is used most commonly in LCA studies of livestock products (De Vries and De Boer, 2010).

Environmental impacts of the following processes in the pig chain were considered and explained below: piglet production (rearing), feed production, manure management, pig housing, and enteric fermentation from pigs (Figure 2).

Environmental impact related to piglet production

Piglet production is defined as the sum of rearing gilts and sows and their piglets that are needed for the production of finishing-pigs (70 days of age, 23.6 kg). In the Netherlands, a sow produces on average 29 weaned piglets per year (Agrovision, 2012). The mortality rate of weaned piglets is 2.2%, whereas the replacement rate of sows is 45%. To replace one culled sow annually we need 0.49 gilt (including death rate).

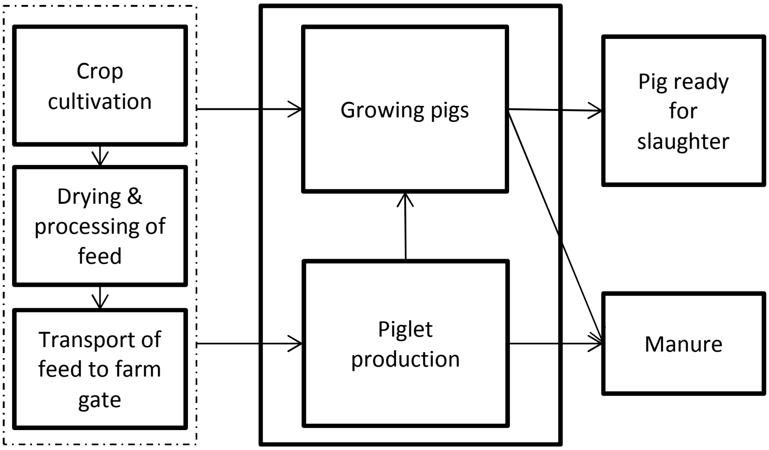

Figure 2. Production chain of finishing-pigs.

Environmental impact related to feed production

GWP, EU, and LU related to feed production were based on Vellinga et al. (2013). Production of feed ingredients included impacts from cultivation (e.g. fertilizers, pesticides, machinery, energy, direct and indirect N_2O emissions, and CO_2 emissions from liming and urea fertilization), impacts from drying/processing, and impacts from transport to the farm. Emissions from LUC were excluded. In the Appendix the environmental impact per kg feed ingredient per diet are described and the diet composition for piglets, gilts, and sows (Appendix Table A.2, A.3, A.4, and A.5).

Environmental impact related to manure management

Handling and storage of manure causes emissions of CH_4 and N_2O. For CH_4, a tier 2 approach was used based on country specific data of Coenen et al. (2013) and Intergovernmental Panel on Climate Change (IPCC) default values (IPCC, 2006). Direct N_2O emissions and indirect N_2O emissions were computed using a tier 2 approach, based on country specific data of Coenen et al. (2013) and IPCC default values (IPCC, 2006). For detailed calculations on the manure emission please consult Appendix Table A.6.

Environmental impact related to pig housing

The environmental impact related to housing is 62 kg CO_2-eq, 689 MJ, and 12.6 m^2 per finishing-pig place per year (EcoinventCentre, 2007). For piglets, gilts, and sows we compensated for the difference in m^2 used per animal place in comparison with the m^2 used per finishing-pig place, based on Dutch regulations (Staatsblad, 2014).

Environmental impact related to enteric fermentation

Enteric methane emission from pigs was calculated using a emission factor of 1.5 kg CH_4 per pig per year (IPCC, 2006).

2.4 Sensitivity analysis

Methodological choices in LCA studies can have a significantly impact on the results. We, therefore, performed a sensitivity analysis to evaluate the robustness of our results. As according to literature, production of feed and manure management explain the majority of GWP, EU, and LU along the life cycle of finishing-pigs, (Basset-Mens and Van der Werf, 2005; Dalgaard et al., 2007) we, therefore, focused on those processes. The GHGs from feed production are merely determined by emissions from land use change (LUC) (Meul et al., 2012; Van Middelaar et al., 2013) and the feed conversion ratio (kg feed intake/kg growth of pigs), which is partly determined by pig characterization in INRAporc. During a sensitivity analysis we, therefore, explored the impact of including LUC emissions, changed the parameters to characterize pig growth, and used a different method to calculate emissions from manure management.

Emissions from LUC

LUC relates to the conversion of land (forest or shrubland) into cropland used for feed production. Calculation methods for LUC emissions show high uncertainty and variability (Meul et al., 2012; Van Middelaar et al., 2013). We, therefore, used two methods: one related to direct LUC and one related to indirect LUC.

The first method focused on direct LUC and attributes the conversion of land in a specific country or region directly to one or more feed ingredients (Jungbluth et al., 2007; Prudêncio da Silva et al., 2010). Soybeans and palm kernel were the only ingredients related to direct LUC. Soybean meal was included in diets of finishing-pig, sows, gilts, and piglets; heat treated soybeans were included in piglet diets; soybean hulls were included in sow diets; and palm kernel expeller was included in sow diets. We assumed that all soy came from Brazil. Soy from South Brazil does not contribute to LUC and from the soy 70% was cultivated in central West Brazil (Prudêncio da Silva et al., 2010). From Central West Brazil, 1% of the soy was assumed to contribute to deforestation of tropical forest, and 3.4% to conversion of shrubland (Prudêncio da Silva et al., 2010). For palm kernel expeller from Malaysia, 100% was assumed to contribute to deforestation of tropical forest (Jungbluth et al., 2007).

Emissions for soy were 825 t CO_2-eq per ha of tropical forest and 297 t CO_2-eq per ha of shrubland and for palm kernel expeller 497 t CO_2-eq per ha (Van Middelaar et al., 2013). An amortization period of 20 years was used. Per kg of SBM LUC emissions were 0.205 g CO_2-eq in addition to 0.652 g CO_2-eq per kg of SBM, per kg of heat treated soybeans LUC emissions were 0.260 g CO_2-eq in addition to 0.663 g CO_2-eq per kg of heat treated soybeans, per kg of soybean hulls LUC emissions were 0.109 g CO_2-eq in addition to 0.373 g CO_2-eq per kg of soybean hulls, and per kg of palm kernel expeller LUC emissions were 0.370 g CO_2-eq in addition to 0.547 g CO_2-eq per kg of palm kernel expeller.

The second method focused on indirect land use. Audsley et al. (2009) state that every ha of land used for commercial production is responsible for total worldwide LUC because food and feed markets are globally interconnected. Thus, total GHG emissions from deforestation at world level in 2004 were divided by the total amount of agricultural land, resulting in one emission factor of 1.43 t CO_2-eq per ha of land.

Characterization of finishing-pigs

The parameters used in INRAporc to characterize a pigs, such as the mean PD, influence the feed conversion ratio. Thus, the feed conversion ratio affects the environmental impact. In a sensitivity analysis we varied these characterization parameters to test whether results between scenarios changed. We based the parameters characterization for the sensitivity analysis on two examples described by Van Milgen et al. (2008). Those two examples were chosen as pigs largely differed in their characterization parameters and, therefore, differences in growth and feed intake were expected. The following input parameters were used: precocity for example one was 0.01 and 0.025 for example two and mean PD was 113 g per day for example one and 179 for example two. Factor a was 1.720 for example one and 2.695 for example two and factor b was 0.606 for example one and 0.577 for example two.

Emissions from manure management

Emissions from manure management were calculated using IPCC default values and average country data. The amount of N excreted by pigs or emission of CH_4 from manure might, however, differ between scenarios, because of differences in diet composition. To analyse a possible impact of diet composition on manure emissions, we calculated N excretion and CH_4 excretion more precisely per scenario.

N-excretion in manure originates from indigestible crude protein in feed ingredients excreted in faeces and the digested crude protein within the urine (urea and uric acid). The N-excretion depends on the N-intake in feed (feed intake multiplied by the crude protein of the feed) minus the N-retention in the animal. The N-retention of the animal is determined by the PD from the INRAporc model.

To calculate CH_4 production a mathematical model (MESPRO) was used (Aarnink et al., 1992). This model quantifies the influence of different diet compositions, feed, and water intake on the manure composition of finishing-pigs. The CH_4 (biogas) production results from anaerobic digestion of manure. Pig diet composition, feed, and water intake can lead to changes in organic matter of the manure, thus influencing biogas production.

3 Results

3.1 Global warming potential

Results expressed per kg body weight, showed that replacing SBM with RSM marginally reduce GWP, less than one percent (Table 3). Expressed per kg of body protein, a reduction of two percent was found in S3 compared with S1. This reduction in S3 is due to an increase in protein to lipid ratio of the pig (relatively high protein content versus lipid content). For S1, S2, and S4 the ratio between protein and lipid content were similar and, therefore, did not lead to different results compared to the results expressed per kg body weight. Feed production for the finishing-pigs had the largest contribution in all scenarios (50-52%), followed by feed production for piglet production (17-18%), manure of finishing-pigs (14%), housing of finishing-pigs (7%), housing related to piglet production (4%), enteric fermentation from finishing-pigs (4%), manure from piglet production (2%), and enteric fermentation related to piglet production (1%).

3.2 Energy use

Results expressed per kg body weight showed that replacing SBM with RSM decreased EU with 1.4% for S2, 2.3% for S3, and 0.4% for S4 (Table 4). Expressed per kg of body protein, again only S3 showed a reduction of three percent EU compared with S1 (due to the different protein-lipid ratio of S3). For all scenarios feed production for finishing-pigs had the largest impact on EU (60-61%), followed by feed production related to piglet production (23%), the housing of finishing-pigs (10%), and housing related to piglet production (6%).

3.3 Land use

Results expressed per kg body weight showed that replacing SBM with RSM decreased LU with 8.6% for S2, 10.3% for S3, and 12.5% for S4 (Table 5). Expressed per kg of protein, again only S3 showed a reduction of 11% LU compared with S1 (due to the different protein-lipid ratio of S3). For all scenarios feed production for finishing-pigs had the largest impact on LU (77-80%), followed by production of feed related to piglet production (19- 21%), the housing of finishing-pigs (1%), and housing related to piglet production (<1%).

Table 3. Impact on GWP (kg CO_2-eq) of replacing soybean meal (S1) with rapeseed meal, based on different diet compositions, and nutritional levels (S2, S3, and S4). The impact per finishing-pig is shown for each production process (e.g. feed). Moreover, the total impact is expressed per kg body weight and body protein.

	Scenario 1	Scenario 2	Scenario 3	Scenario 4
Diet NE content, MJ/kg	9.50	8.98	8.98	9.50
SID LYS g/kg	7.59	7.18	7.18	7.59
Impact per finishing-pig, kg CO_2-eq				
Piglet production	69.5	69.5	69.5	69.5
Feed	148.4	148.2	141.4	147.2
Manure	39.6	39.6	39.6	39.6
Housing	19.3	19.3	19.3	19.3
Fermentation	11.3	11.3	11.3	11.3
Impact per kg body weight	2.5	2.5	2.5	2.5
Impact per kg protein	15.0	15.0	14.8	15.0

Table 4. Impact on energy use (MJ) of replacing soybean meal (S1) with rapeseed meal, based on different diet compositions, and nutritional levels (S2, S3, and S4). The impact per finishing-pig is shown for each production process (e.g. feed). Moreover, the total impact is expressed per kg body weight and protein.

	Scenario 1	Scenario 2	Scenario 3	Scenario 4
Diet NE content, MJ/kg	9.50	8.98	8.98	9.50
SID LYS g/kg	7.59	7.18	7.18	7.59
Impact per finishing-pig, MJ				
Piglet production	600.5	600.5	600.5	600.5
Feed	1 293.6	1 265.1	1 207.0	1 284.5
Housing	213.3	213.3	213.3	213.3
Impact per kg body weight	18.1	17.9	17.7	18.0
Impact per kg protein	109.9	108.5	106.1	109.5

Table 5. Impact on land use (LU in m²) of replacing soybean meal (S1) with rapeseed meal, based on different diet compositions and nutritional levels (S2, S3, and S4). The impact per finishing-pig is shown for each production process (e.g. feed). Moreover, the total impact is expressed per kg body weight and protein.

	Scenario 1	Scenario 2	Scenario 3	Scenario 4
Diet NE content, MJ/kg	9.50	8.98	8.98	9.50
SID LYS g/kg	7.59	7.18	7.18	7.59
Impact per finishing-pig, m^2				
Piglet production	97.5	97.5	97.5	97.5
Feed	409.6	365.4	348.7	345.9
Housing	3.9	3.9	3.9	3.9
Impact per kg body weight	4.4	4.0	3.9	3.8
Impact per kg protein	26.7	24.4	23.6	23.3

3

3.4 Sensitivity analysis

Impact of emissions from LUC

Replacing SBM with RSM, while accounting for direct LUC, decreased GWP per kg of body weight with 2.8% for S2, 3.4% for S3, and 3.2% for S4, compared with S1. The absolute impact of GWP increased with 4% for S1, 1% for S2, 1% for S3, and 1% for S4.

Replacing SBM with RSM, while accounting for indirect LUC, decreased GWP with 1.8% for S2, 2.6% for S3, and 2.9% for S4, compared with S1 per kg of body weight. Although differences between scenarios remained marginal, the absolute impact of the GWP increased with 25% for S1, 23% for S2, 23% for S3, and 22% for S4 per kg body weight.

Thus, including emissions related to LUC did not result in differences between scenarios, but the absolute value of each scenarios changed. Therefore, including LUC emissions did not have an impact on the final conclusion.

Impact of pig characteristics

The effect of changing the pig characterization parameters marginally affected the results of GWP, EU, and LU (Table 6). The largest change occurred in example two for LU. LU decreased with 0.7% for S2, 1.0% for S3, and 1.1% for S4 compared with S1 per kg of body weight. The absolute level of LU within each scenarios, however, increased with 11.5% for S1 with 10.4% for S2, 10.1% for S3, and 10.2% for S4 per kg body weight due to changing pig characterization parameters. Thus, the impact of changing pig characterization parameters did not results in differences between scenarios, but the absolute value of each scenarios increased. Therefore, changing pig characterization parameters did not have an impact on the final conclusion.

Impact of emissions from manure management

Concerning manure emissions, compared with S1, GWP increased with 0.3% for S2, while S3 decreased with 0.5%, and S4 with 0.3% expressed per kg of body weight. The absolute impact of the GWP within scenarios decreased with 0.9% for S1, 0.6% for S2, 0.8% for S3, and 0.8% for S4 expressed per kg body weight. Thus the impact on the original results of using a more precise methods to calculate N excretion by the pig and CH_4 production was relatively small between scenarios, even as the absolute impact within scenarios.

Table 6. Impact of sensitivity analysis of characterisation parameters in INRAporc on the scenarios of replacing soybean meal (S1) with rapeseed meal based on different diet compositions and nutritional levels (S2, S3, and S4). Two sets of characterisation parameters were used based on Van Milgen et al., 2008. For each example the body weight (BW), feed intake, and the environmental impact: global warming potential (GWP) expressed in CO_2-eq, energy use (EU) in MJ, and land use (LU) in m^2 per year per kg body weight.

	Scenario 1	**Scenario 2**	**Scenario 3**	**Scenario 4**
NE content, MJ/kg	9.50	8.98	8.98	9.50
SID LYS g/kg	7.59	7.18	7.18	7.59
Original BW	116.4	116.4	114.29	116.4
Example 1	109.4	109.4	107.2	109.4
Example 2	159.1	159.1	156.2	159.1
Original feed intake	226.0	237.0	226.0	226.0
Example 1	217.0	228.0	217.0	217.0
Example 2	374.0	392.0	374.0	374.0
Original GWP	2.5	2.5	2.5	2.5
Example 1	2.6	2.6	2.6	2.6
Example 2	2.4	2.4	2.4	2.4
Original EU	18.1	17.9	17.7	18.0
Example 1	18.8	18.6	18.4	18.7
Example 2	18.6	18.3	18.0	18.4
Original LU	4.4	4.0	3.9	3.8
Example 1	4.5	4.1	4.0	3.9
Example 2	4.9	4.4	4.3	4.2

4 Discussion

A previous review showed a variation between 3.9-10 kg CO_2-eq, between 18-45 MJ EU, and between 8.9-12.1 m^2 LU per kg edible product (De Vries and De Boer, 2010). Our results are within the range of results reviewed by De Vries and De Boer (2010) (GWP between 4.64-4.67 kg CO_2-eq, an EU between 33-34 MJ, and a LU between 7.25-8.28 m^2), although LU is a bit lower. Furthermore, our study support the earlier finding that feed production causes the majority of GWP, EU, and LU (Eriksson et al., 2005; Dalgaard et al., 2007). To gain insight into the full environmental impact of replacing SBM with RSM, and to prevent burden shifting the environmental impacts eutrophication and acidification should be assessed as well.

To our knowledge, no other studies aimed to assess the environmental impact of replacing SBM with RSM in finishing-pig diets, although some assessed the impact of replacing SBM with locally produced protein sources such as peas, lupines, and rapeseed products (Eriksson et al., 2005; Meul et al., 2012; Sasu-Boakye et al., 2014). Eriksson et al. (2005) found a reduction in GWP up to 13% and in EU up to 22%. They concluded that feeding strategies have potential to reduce environmental impacts. Sasu-Boakye et al. (2014) found a reduction in GWP up to 4.5% and 11% for LU. Meul et al. (2012) found a reduction in GWP up to 3%. When accounting for emissions related to direct LUC Meul et al. (2012) found a reduction in GWP up to 15%, whereas accounting for indirect LUC resulted in a reduction of only one percent.

On the other hand our study, indicates that the impact of replacing SBM with RSM is marginal and remains marginal when emissions related to direct (up to 3.4%) and indirect LUC (up to 2.9%) are included. In first instance our results seem to contradict results of Eriksson et al. (2005) and Meul et al. (2012). Differences in result come from differences in impact values used and system boundaries used.

Meul et al., 2012 used impact values of 0.555 kg CO_2-eq and 3.06 m^2 per kg SBM, and 0.437 kg CO_2-eq and 1.14 m^2 per kg RSM. Eriksson et al., 2005 used impact values of 0.73 kg CO_2-eq and 5.02 MJ per kg SBM, and 0.37 kg CO_2-eq and 2.39 MJ per kg RSM. Where we used 0.652 kg CO_2-eq, 3.1 m^2, and 6.1 MJ per kg SBM, and 0.454 kg CO_2-eq, 1.2 m^2, and 3.1 MJ per kg RSM. The relative high reduction found by Eriksson et al. (2005) (reduction in GWP up to 13%), for example, can be explained by the relatively high difference in CO_2-eq per kg, between SBM and RSM.

The system boundaries used by Eriksson et al. (2005) and Meul et al. (2012) also differ from our study. Meul et al. (2012), for example, evaluated the environmental impacts of feed production only, and excluded other processes, such as manure management, piglet production, and pig housing. Eriksson et al. (2005) excluded the environmental impact related to piglet production and enteric fermentation. In case we evaluated environmental impact of replacing SBM with RSM for the process of feed production only, GWP (excluding LUC) decreased from 0.1% to 2.9%, EU decreased from 0.7 to 5.0%, and LU decreased from 10.8% to 15.6%, compared with S1. In case we accounted for emissions related to direct LUC the GWP decreased from 5% to 10%, while accounting for emissions of indirect LUC GWP was decreased from 3% to 8%. The relative importance of replacing SBM with RSM obviously depends on the level of analysis and decreases with including chain processes other than feed production, such as piglet production, manure management, and pig housing. For our study it was essential to evaluate the environmental consequences of replacing SBM with RSM along an extended chain because scenarios evaluated affected final body weight of pigs.

We should, however, note that, there are large difference in the impact of LUC between studies, due to different assumptions related to the percentage of soy expansion in central Brazil in forest and shrubland. We assumed that 1% of the soy produced in Central West Brazil comes from tropical forest, and 3.4% comes from shrubland, whereas soy from South Brazil does not contribute to LUC. We based this on the work of Prudêncio da Silva et al. (2010). In literature, however, the following assumptions were found: Van Middelaar et al. (2013) used the same values; Gerber et al. (2013) assumed that 100% of the soy expansion in Brazil directly occurs on forest land; Nemecek et al. (2014) assumed that 12% of the soy produced in Central West Brazil comes from tropical forest, and 38% comes from shrubland; Meul et al., (2012) assumed that 3% of the soy produced in Central West Brazil comes from tropical forest, and 5% comes from shrubland; Persson et al. (2014) assumed that 2% of the soy produced in Central West Brazil comes from tropical forest, and 12% comes from shrubland. Moreover, based on satellite data it has been shown that since 2006, deforestation rates in Brazil have decreased, and that since the late 2000s the contribution of soy production to deforestation has been minimal (i.e., due to anti-deforestation measures; (Macedo et al. 2012)).

Another discussion point is the amortization period (20 years) we used. Emission of soy per ha of LUC include emissions related to the moment the land is cleared and used for another purpose and emissions related to C-sequestration. It is debatable to amortizes the emission related to the clearing of the land, however, as this is mostly applied in LCA studies e.g. Meul et al. (2012), Van Middelaar et al. (2013), and Nemecek et al. (2014), we used an amortization period of 20 years for both emissions related to the moment the land is cleared and emissions related to C-sequestration.

Similar to the sensitivity results of LUC, changing the methodology to calculate manure emissions hardly affected the relative differences between scenarios. Sensitivity results of changing pig characterization parameters also hardly affected the relative differences between scenarios. A change in characterization parameters, however, in some cases increased or decreased the absolute impact of all scenarios considerably (up to 11.5% for LU). The impact of changing characterization parameters on the environmental impact can be explained by the fact that it influences the relative rate of PD and LD. PD follows a curvilinear plateau function, in response to energy supply (Van Milgen et al., 2008). When PD attains the plateau, all additional feed energy is used for LD, which increases linearly with energy intake. Compared to PD, however, it requires more feed to gain one kg of LD. Changing pig characterization parameters, therefore, affects the balance between PD and LD resulting in differences in feed conversion ratio. This balance between PD and LD also explains why scenario S3 had most potential to reduce the environmental impact. Besides the fact that S3 had a low environmental impact per kg of feed, the reduced feed intake changed the relative rate between PD and LD. So, pigs in S3 had a higher protein to lipid ratio resulting in a better feed conversion ratio compared to S1, S2, and S4. Optimizing this relative rate of protein and lipid deposition by changing genetic characterization parameters and managing feed intake, therefore, might result in an improved absolute environmental impact of pig production.

5 Conclusions

Results show that, expressed per kg of body weight, replacing SBM with RSM in diets of finishing-pigs did not result in a different GWP or EU, whereas LU decreased up to 12%. Between scenarios, S3 had most potential to reduce the environmental impact, especially when the impact was expressed per kg of protein mass. Besides the fact that S3 had a low environmental impact per kg of feed, the reduced feed intake changed the relative rate between PD and LD. So, pigs in S3 had a higher protein to lipid ratio resulting in a better feed conversion ratio compared to S1, S2, and S4. Optimizing this relative rate of PD and LD by diet composition, feed allowance, and genetic characterization parameters, therefore, might result in an improved absolute environmental impact of pig production. Furthermore, it was found that the impact of replacing SBM with RSM in diets of finishing-pigs per kg of body weight changed marginally when emissions related to direct (up to 3.4%) and indirect LUC (up to 2.9%) were included. In case we evaluated environmental impacts of feed production only, which implies excluding other processes along the chain as is generally found in literature, GWP decreased up to 10% including LUC, EU up to 5%, and LU up to 16%.

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Appendix

Table A.1 describes the nutritional composition of the four scenarios for finishing-pigs.

Table A.1. Nutritional composition of the four scenarios of replacing soybean meal (SBM) (S1) with rapeseed meal (RSM) based on different diet compositions (S2, S3, and S4). See Table 1 in the article for the diet composition.

	Scenario 1	**Scenario 2**	**Scenario 3**	**Scenario 4**
	SBM	RSM	RSM	RSM
Net energy, MJ	9.50	8.98	8.98	9.50
Lysine (SID)	7.59	7.18	7.18	7.59
Methionine (SID)	2.53	2.45	2.45	2.53
Cysteine (SID)	2.41	2.75	2.75	2.41
Threonine (SID)	4.88	4.73	4.73	4.88
Tryptophan (SID)	1.55	1.42	1.42	1.55
P	3.75	4.84	4.84	4.65
Dig. P	2.27	2.14	2.14	2.27
Crude protein	162.00	163.00	163.00	160.00
Crude fat	27.00	31.00	31.00	50.00
Crude fibre	30.00	49.00	49.00	47.00

Table A.2 provides information on the environmental impact of feed ingredients based on Vellinga et al. (2013) and the diet composition of piglets, gilts, and sows diets is given.

Table A.2. Diet composition of piglets, gilts, and sows and the related environmental impact.

Ingredient %	Piglets	Gilt	Sow	GWP g CO_2-eq/kg	EU MJ/kg	LU m^2/kg
Animal fat	2.00	2.00	0.70	823	12.0	0.00
Barley	32.10	6.78	13.67	375	2.7	1.30
Bread meal	1.75	2.25	1.58	125	1.9	0.00
Palm kernel expeller	-	-	3.25	547	3.2	0.30
Sunflower oil	0.50	0.41	0.49	2158	21.0	16.29
Potato protein	1.35	-	-	1798	20.0	1.70
Maize	21.50	25.00	21.13	704	6.3	1.20
Rapeseed expeller	-	-	0.84	501	3.3	1.40
Rapeseed meal	-	10.00	1.30	454	3.1	1.20
Soybean hulls	-	2.54	6.00	373	3.9	1.60
Soybeans heat treated	0.11	-	-	663	5.8	3.90
Soybean meal	12.90	4.25	3.70	652	6.0	3.10
Sunflower seed meal	3.00	5.00	3.49	519	5.4	3.10
Sugar beet-pulp	1.00	5.00	8.82	366	5.6	0.00
Sugarcane molasses	1.44	2.00	2.40	302	3.7	0.22
Triticale	-	1.13	-	564	3.7	1.15
Wheat	10.99	20.40	12.35	367	2.7	1.10
Wheat middlings	5.00	10.15	15.73	237	2.0	0.58
Whey powder	1.00	-	-	1016	15.0	0.00
DL-Methionine	0.14	0.02	0.01	5490	89.3	0.01
Lactic Acid	1.00	-	-	4999	75.2	0.11
Limestone	1.00	0.90	1.11	19	0.0	0.00
L-Lysine	0.45	0.30	0.24	6030	119.9	2.27
L-Threonine	0.12	0.05	0.07	16978	119.9	2.27
Phytase	0.65	0.65	0.65	4999	26.0	0.15
Premix	0.40	0.40	0.40	4999	0.9	0.00
Monocalcium phosphate	0.66	0.11	0.38	4999	18.4	0.32
Salt	0.60	0.30	0.54	180	3.9	0.02

Table A.3 shows the diet composition with the related environmental impact per feed ingredient for piglets, gilts, and sows. The diet for piglets, gilts, and sows is composed using least cost optimization according to the procedure described by Bikker et al. (2011) using data and costs of ingredients in 2013. The feed intake of piglets was 30 kg per piglet, 403 kg per gilt, and 1174 kg per sow per year (Agrovision, 2012). The diets of piglets contained 9.68 MJ NE, for gilts 9.24 MJ NE, and 9.06 MJ NE for sows. The environmental impact related to transporting the feed to the feed mill is 58 g CO_2-eq per kg feed and 0.82 MJ per kg feed for piglets, 61 g CO_2-eq per kg feed and 0.87 MJ per kg feed for gilts, and 50 g CO_2-eq and 0.71 MJ per kg feed for sows. The environmental impact related to transporting the feed to the farm is 10 g CO_2-eq per kg feed and 0,15 MJ per kg feed for piglets, gilts, and sows.

Table A.3. Environmental impact of feed ingredients for finishing-pigs.

Ingredients	**GWP**	**EU**	**LU**
	g CO_2-eq/kg	MJ/kg	m^2/kg
Rapeseed meal	454	3.1	1.2
Soybean meal	652	6.0	3.1
Peas	731	6.4	5.5
Maize	704	6.3	1.2
Wheat	367	2.7	1.1
Wheat middlings	237	2.0	0.6
Barley	375	2.7	1.3
Sugarcane molasses	302	3.7	0.2
Phytase	4999	26.0	0.2
Premix	4999	0.9	0.0
Animal fat	823	12.0	0.0
Limestone	19	0.0	0.0
Salt	180	3.9	0.0
Monocalcium phosphate	4999	18.4	0.3
L-Lysine	6030	119.9	2.3
L-Threonine	16978	119.9	2.3
DL-Methionine	5490	89.3	0.0
Transport to feed mill	47	0.7	0.0
Transport to farm	10	0.2	0.0

Table A.4 provides for each scenario an overview of GWP, LU, and EU per kg of feed.

Table A.4. Global warming potential (GWP), land use (LU), and energy use (EU) per kg feed. Four scenarios were developed in which soybean meal (scenario 1) was replaced with rapeseed meal (scenario 2, 3, and 4) in diets of finishing-pigs.

		Scenario 1	Scenario 2	Scenario 3	Scenario 4
Diet NE content, MJ/kg		9.50	8.98	8.98	9.50
SID LYS g/kg		7.59	7.18	7.18	7.59
GWP	g CO_2-eq/kg feed	656	626	626	651
EU	MJ/kg feed	5.72	5.34	5.34	5.68
LU	m^2.year / kg BW	1.81	1.54	1.54	1.53

Table A.5 provides for piglets, gilts, and sows an overview of GWP, LU, and EU per kg of feed.

Table A.5. Global warming potential (GWP), land use (LU), and energy use (EU) per kg feed for piglet production. Diets of piglets, gilts, and sows were based on Dutch standards.

		Piglet	Gilt	Sow
GWP	g CO_2-eq/kg feed	756	625	593
EU	MJ/kg feed	7.32	5.87	5.47
LU	m^2.year / kg BW	1.45	1.22	1.11

Table A.6 shows the equations to calculate CH_4, direct and indirect N_2O emissions with the related parameters. Finishing-pigs emitted 3.56 kg CO_2-eq related to direct N_2O, 4.45 kg CO_2-eq related to indirect N_2O, and 31.66 kg CO_2-eq related to CH_4. The emission related to piglet production was 1.21 kg CO_2-eq related to direct N_2O, 1.51 kg CO_2-eq related to indirect N_2O, and 3.26 kg CO_2-eq related to CH_4.

Table A.6. Equations to calculate CH_4, direct and indirect N_2O emissions with the related parameters.

Parameters		**Finishing pigs**	**Gilts**	**Sows**	**References**
CH_4 emissions = $(OM * BO * 0.67 * MCF) * amount\ manure$					
Organic matter (OM)	kg/OM manure	43	25	25	Coenen et al., 2013
Potential CH_4 production (B0)	CH_4/kg OM	0.34	0.34	0.34	Coenen et al., 2013
CH_4 conversion factor (MCF)		0.39	0.39	0.39	IPCC, 2006
Manure production	kg/year	1100	1300	5100	Coenen et al., 2013
Direct N_2O emissions = $EF * N * (\frac{44}{28})$					
N excretion	kg/year	12.5	15.9	30.1	Coenen et al., 2013
Default emission factor (EF)		0.002	0.002	0.002	IPCC, 2006
Indirect N_2O emissions = $Volatilisation * EF * (\frac{44}{28})$					
N excretion	kg/year	12.5	15.9	30.1	Coenen et al., 2013
Volatilisation	%	25	25	25	IPCC, 2006
Default emission factor (EF)		0.01	0.01	0.01	IPCC, 2006

Chapter 4

Assessing environmental consequences of using co-products in animal feed

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Abstract

Purpose. The livestock sector has a major impact on the environment. This environmental impact may be reduced by feeding agricultural co-products (e.g. beet tails) to livestock, as this transforms inedible products for humans into edible products, e.g. pork or beef. Nevertheless, co-products have different applications such as bio-energy production. Based on a framework we developed, we assessed environmental consequences of using co-products in diets of livestock, including the alternative application of that co-product.

Methods. We performed a consequential life cycle assessment, regarding greenhouse gas emissions (including emissions related to land use change) and land use, for two case studies. Case 1 includes increasing the use of wheat middlings in diets of dairy cattle at the expense of using it in diets of pigs. The decreased use of wheat middlings in diets of pigs was substituted with barley, the marginal product. Case 2 includes increasing the use of beet tails in diets of dairy cattle at the expense of using it to produce bio-energy. During the production of biogas, electricity, heat, and digestate (that is used as organic fertilizer) were produced. The decrease of electricity and heat was substituted with fossil fuel, and digestate was substituted with artificial fertilizer.

Results and discussion. Using wheat middlings in diets of dairy cattle instead of using it in diets of pigs resulted in a reduction of 329 kg CO_2-eq per ton wheat middlings and a decrease of 169 m^2 land. Using beet tails in diets of dairy cattle instead of using it as a substrate for anaerobic digestion resulted in a decrease of 239 kg CO_2-eq per ton beet tails and a decrease of 154 m^2 land. Emissions regarding land use change contributed significantly in both cases but had a high uncertainty factor, ±170 ton CO_2 ha^{-1}. Excluding emissions from land use change resulted in a decrease of 9 kg CO_2-eq for case 1 'wheat middlings' and an increase of 50 kg CO_2-eq for case 2 'beet tails'.

Conclusions. Assessing the use of co-products in the livestock sector is of importance because shifting its application can reduce the environmental impact of the livestock sector. A correct assessment of the environmental consequences of using co-products in animal feed should also include potential changes in impacts outside the livestock sector, such as the impact in the bio-energy sector.

1 Introduction

Current livestock production levels pose severe pressure on the environment via their emissions to air, water, and soil (Tilman et al., 2001; Steinfeld et al., 2006). The livestock sector also competes increasingly for scarce resources such as land, water, and fossil-energy (Steinfeld et al., 2006; De Vries and De Boer, 2010). The challenge, therefore, is to reduce emissions to the environment and to increase efficient use of scarce resources per kilogram of animal-source food produced.

The environmental impact of livestock production results mainly from production and utilization of feed (Van der Werf et al., 2005; Elferink et al., 2008; De Vries and De Boer, 2010). A possible way to reduce the impact of livestock production is feeding co-products from, for example, arable production or the food processing industry to livestock. Examples of co-products are wheat middlings, a co-product from wheat cultivated to produce wheat flour, or beet tails, a co-product from sugar beets cultivated to produce sugar. As most emissions or resources used, for example during crop cultivation or processing, are ascribed to the main product that economically drive these production stages, the environmental impact of an untreated co-product is according to Elferink et al. (2008) relatively low. Furthermore, most co-products are inedible for humans or do not meet Dutch food requirements, such as taste and texture. Therefore, feeding co-products to livestock transforms an inedible product into an edible product, such as meat, milk, and eggs (Fadel, 1999; Elferink et al., 2008; Garnett, 2009).

Some co-products are used in diets of livestock (Nonhebel, 2007; Elferink et al., 2008; Vellinga et al., 2009). Vellinga et al. (2009) showed that in the Netherlands in 2007, the amount of co-products used in diets of livestock was 22%. The current motivation to use co-products in diets of livestock, however, depends on a combination of their nutritional value and cost price, and is not driven by environmental motives. Elferink et al. (2008) concluded that for all Dutch citizens about 81 g of pork per day can be produced, while using all co-products from the sugar beet, vegetable oil, and potato industry, which represent approximately 60 % of the co-products produced from the food industry in the Netherlands. When corrected for the total share of co-products in feed produced in the Netherlands, enough pig meat can be produced to fulfil the amount of animal protein advised by the Dutch health organizations, while the environmental impact per kilogram of meat produced decreased (Elferink et al., 2008). However, we should take into account that, besides feed, co-products might have other applications, such as production of bio-energy, or can be fed to other species. Increasing the use of a co-product in animal feed inherently implies decreasing the availability of that co-product for other applications as the production of the co-product is determined by the main product that economically drives the production stages. In the Netherlands, for example, there is a competition between animal feed and bio-energy production for wet co-products, such as beet tails and potato peels (Koppejan et al., 2009). Increasing the use of co-products in the livestock sector, therefore, may have an environmental impact on processes outside the production cycle of the livestock sector, e.g. on processes in the bio-energy industry. That impact needs to be considered when evaluating the environmental impact of using co-products in the livestock sector.

The goal of this paper was to assess the overall environmental consequences of increasing the use of co-products in diets of livestock, including environmental consequences for the alternative application of that co-product. We used consequential life cycle assessment (CLCA) to illustrate the overall consequences for two Dutch case studies, regarding global warming potential (GWP) and land use (LU). In the first case, we analysed the consequences of increasing the use of wheat middlings in diets of dairy cattle at the expense of using it in diets of pigs, whereas in the second case, we analysed the consequences of increasing the use of beet tails in diets of dairy cattle at the expense of using it to produce bio-energy. These cases were of interest as both co-products are used as energy source in livestock feed and, therefore, are comparable with respect to feed requirements. Furthermore, both co-products are used as dairy cattle feed but they differ in their alternative application. Wheat middlings are used in pig feed but can be used in dairy cattle feed as well. Beet tails are used in the bio-energy sector (De Vries et al., 2012b) but can be used in dairy cattle feed as well.

2 Methods

2.1 Consequential life cycle assessment

Life cycle assessment (LCA) is an internationally accepted and standardized holistic method (ISO14040, 1997; ISO14041, 1998; ISO14042, 2000; ISO14043, 2000) to evaluate the environmental impact during the entire production chain (Guinée et al., 2002; Bauman and Tillman, 2004). In this study, we focused on LU and GWP (including emissions from land use change (LUC)).

Two types of LCA exist: attributional LCA and CLCA. Attributional LCA describes the environmentally relevant physical flows to and from a product or process, while CLCA describes how environmental flows change in response to a change in the system (Ekvall and Weidema, 2004). As our aim was to determine environmental consequences of a change in use of co-products, we performed a consequential LCA.

The starting point in our CLCA was a multifunctional process[1], an activity that fulfils more than one function (Ekvall and Finnveden, 2001) yielding two products: the determining product, which determines the production volume of that process, and a co-product (Weidema et al., 2009). A change in demand of the determining product directly affects the production volume of the co-product, and subsequently the production of the product that is displaced by that co-product (Weidema et al., 2009). Within CLCA, system expansion is generally used to deal with multifunctional processes. System expansion implies that you include changes in the environmental impact of the alternative production process, for which the co-product could be used, into your analysis by subtracting the impact related to the alternative production process[2] (Ekvall and Finnveden, 2001). Weidema et al. (2009) developed a framework to ascribe the environmental impact of a multi-functional process to various outputs, based on system expansion. Their framework, however, was based on the assumption that the demand for the determining product or co-product increased or, in other words, the total amount of available co-product increased. In our analysis, however, we assumed a stable market situation. We wanted to assess the consequences of changing the application of a co-product, while the demand remained equal. When the demand for both the determining product and co-product remained equal, the total amount of available co-product remained equal as well. In this way, we could analyse what the optimal use of a co-product is from an environmental perspective. Therefore, we extended the framework of Weidema et al. (2009) to allow an analysis of the environmental consequences of a change in application of a co-product, for example, a change from application in bio-energy production to application in diets of livestock. The extended framework is explained in Section 2.4.

2.2 Case description

Two co-products, i.e. wheat middlings and beet tails, were selected as cases to illustrate our extended framework.

Case 1: wheat middlings

Milling of wheat results in the production of wheat flour used for human consumption (the determining product) and wheat middlings (the co-product). The production volume of wheat middlings, therefore, is determined by the demand for wheat flour.

In this case study, we illustrated the environmental consequences of increasing the use of wheat middlings with one ton in diets of dairy cattle while the number of animals remained

[1] Also referred to as product-package in this thesis.

[2] Also referred to as net environmental impact in this thesis.

equal. We assumed that an increased use of wheat middlings in diets of dairy cattle inherently implied a decreased use of wheat middlings in diets of pigs. Wheat middlings are used in dairy cattle feed and pig feed for their energy content. The decreased use of wheat middlings in diets of pigs must be substituted with an alternative product while the number of pigs remained equal. The marginal energy-rich fodder was assumed to be barley produced in the Netherlands (Weidema, 2003). Wheat middlings and barley are both products with a high energy and low protein content.

Case 2: beet tails

Beet tails (the co-product) are cut off after first cleaning (screening and washing) of sugar beets during the production of sugar (the determining product). The production volume of beet tails, therefore, is determined by the demand for sugar.

In this case study, we illustrated the environmental consequences of increasing the use of beet tails with one ton in dairy cattle feed while the number of animals remained equal. Beet tails are used in dairy cattle feed for their energy content, but can alternatively be used for production of bio-energy. During the conversion of biomass by anaerobic digestion into biogas, methane (CH_4), carbon dioxide (CO_2), and trace gases (e.g. hydrogen gas) are produced, which can be used to produce bio-energy in the form of electricity, heat, or transport fuel (Hamelin et al., 2011; De Vries et al., 2012a). The remaining product after digestion is called 'digestate' and can be used as organic fertilizer replacing artificial fertilizer (Börjesson and Berglund, 2007). We assumed that an increased use of beet tails as dairy cattle feed inherently implied a decreased use of beet tails for bio-energy production. The decreased production of electricity, heat, and digestate must be substituted with an alternative product, i.e. the marginal product. Electricity was assumed to be substituted with marginal Dutch electricity, i.e. 28% coal-based, 67% natural gas-based, and 5% wind-based electricity (De Vries et al., 2012a). Fifty percent of the heat was assumed to be substituted with marginal heat, i.e. 79% natural gas-based and 21% light fuel oil-based in the Netherlands. The rest is used for digestion processes and, therefore, no alternative products were included (De Vries et al., 2012a). The digestate that is transported and applied to the field as fertilizer was assumed to be substituted by marginal mineral N, P, and K fertilizer. Marginal production of mineral fertilizer was assumed to be calcium ammonium nitrate for N, triple superphosphate for P_2O_5, and potassium chloride for K_2O (De Vries et al., 2012a).

2.3 Environmental consequences

We assessed the consequences of a change in co-product use for greenhouse gas (GHG) emissions and LU. Emission of GHGs and LU were chosen as an example as the livestock sector has a significant contribution to both climate change and LU worldwide (Steinfeld et al., 2006). Emissions of GHGs regarding LUC were included, but reported separately. The following GHGs were included: CO_2, CH_4, and nitrous oxide (N_2O). We assessed the change in global warming potential per ton co-product, i.e. wheat middlings or beet tails, by summing up changes in emissions of these GHGs based on their equivalence weighting factors in terms of CO_2 (100 years' time horizon): i.e. 1 for CO_2, 25 for CH_4 and 298 for N_2O (Forster et al., 2007).

GHG emissions associated with the production of beet tails and barley were based on data of De Vries et al. (2012b). LUC and LU data related to the cultivation of barley were based on Tonini et al. (2012) and De Vries et al. (2012b). When computing LUC, we focused on the cultivation of barley only and excluded low land use processes such as transport. Tonini et al. (2012) quantified CO_2 emissions of converting, for example, forest or grassland to cropland, accounting for size and location of converted land and the types of land that were converted (biome types). De Vries et al. (2012b) assumed that 1.22 ha of land needed to be converted somewhere in the world to compensate for the use of 1 ha (average Dutch yield) of barley in the Netherlands. A LUC emission factor of 310 ton CO_2 ha^{-1} of displaced barley was derived, with an uncertainty of ±170 ton CO_2 ha^{-1} (Tonini et al., 2012). This corresponds to 1.55 kg CO_2 m^{-2} $year^{-1}$ (±0.84 kg CO_2 m^{-2} $year^{-1}$) with an amortization period of 20 years (as prescribed in the Renewable Energy Directive (EU, 2009).

2.4 Framework

Figure 1 illustrates our extended framework based on Weidema et al. (2009). The terminology used in this chapter and figures were based on Weidema et al. (2009), as we extended their framework.

The multifunctional process was denoted as process A, where product A was the determining product. The process described the environmental impact related to the product. The process is referred to in italics. *Process B* was the process related to the use of the co-product. The intermediate process (*process I*) was a process or series of processes between the point where the co-product left the process route of the determining product and its use in *process B*. The product produced during this intermediate process was defined as intermediate product (product I). In case product I was not available for *process B*, another product, i.e. product D, was used. Use of product I in *process B*, therefore, displaced use of product D. The difference in environmental impact in *process B* due to using product I instead of product D was denoted by ΔB. If a co-product was not fully used, it went to waste treatment, *process W*.

Figure 1 shows the environmental consequences of three possible changes in application of a co-product. These three situations are explained below.

Situation 1: changing the application of a co-product

Situation 1, i.e. changing the application of a co-product from *process B1a* to *B1b*, corresponds with the first case: wheat middlings fed to dairy cattle instead of pigs. The environmental impact of this change in application is determined as:

D1a-ΔB1a-D1b+ΔB1b. (1)

Parameters are explained in the following paragraphs.

Computing *D1a* and *D1b*

D1a is the environmental impact related to the production of product D1a that is needed to replace product I in *process B1a*. To quantify *D1a* in our case of wheat middlings, we needed to determine the amount of barley required to replace 1 kg of wheat middlings in pig feed, and the environmental impact and LU of producing 1 kg of barley. Wheat middlings and barley were assumed to be exchanged on the basis of their net energy content. The available net energy in pig feed is expressed in EW (in Dutch, Energie Waarde (energy value)). The EW of wheat middlings and barley were obtained from the feed tables of the Dutch Central Bureau for Livestock Feeding (CVB, 2010). Wheat middlings contain 0.75 EW per kg, whereas barley contains 1.05 EW per kg. To replace 1 kg of wheat middlings in pigs feed, we need 0.71 kg barley. Given a dry mater (DM) content of barley of 86.9%, 621 kg of DM barley was needed to replace 1 ton of wheat middlings (i.e. 0.71 kg barley×0.869×1000=621 kg). The production of 1 kg DM barley results in 0.44 CO_2 eq (excl. LUC) and a land use of 1.60 m^2 (De Vries et al., 2012b).

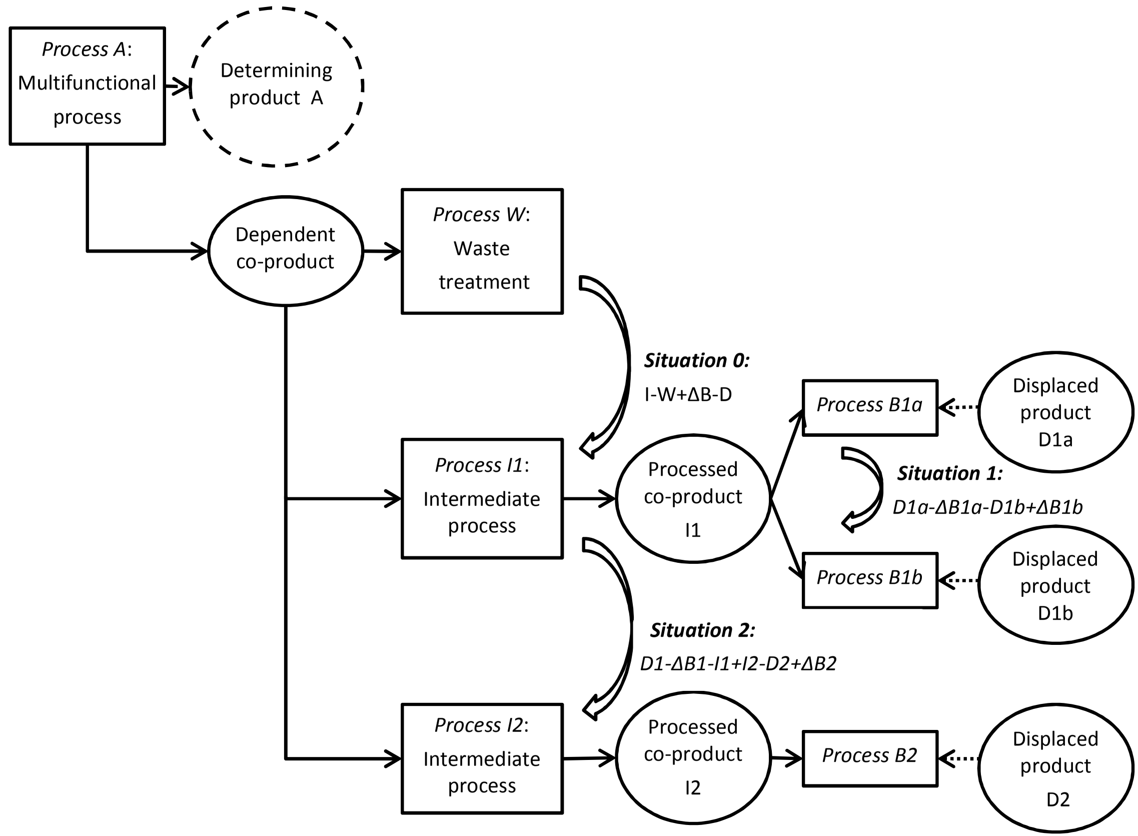

Figure 1. Framework for assessing environmental consequences of changing the application of co-products in diets of livestock while the production of product A and B remains the same, three different situations (based on Weidema et al., 2009).

D1b is the environmental impact related to the production of product D1b that is avoided because of the use of product I in *process B1b*. To quantify *D1b* in our case of wheat middlings, we needed to determine the amount of barley avoided per kilogram of wheat middlings in dairy cattle feed, and the environmental impact and LU of 1 kg of barley (similar as *D1a*). Just like for pigs, barley and wheat middlings were assumed to be exchanged in diet of dairy cattle on the basis of their net energy content. In the Netherlands, the available net energy in dairy cattle feed is expressed in VEM (In Dutch, Voeder Eenheid Melk (fodder unit milk). Wheat middlings contains 815 VEM per kg, whereas barley contains 975 VEM per kg (CVB, 2010). To replace 1 kg of wheat middlings in diets of dairy cattle, we needed 0.84 kg of barley. Given a DM content of barley of 86.9%, 726 kg of DM barley was needed to replace 1 ton of wheat middlings (0.84 kg barley×0.869×1000=726 kg).

Computing *ΔB1a* and *ΔB1b*

ΔB1a is the difference in environmental impact of *process B1a*, i.e. feed utilization of a pig when feeding wheat middlings instead of barley. During the digestive process, ruminants, and to a minor extent monogastric animals, emit CH_4 (Steinfeld et al., 2006). Changing the diet to a higher fibre composition can increase the enteric fermentation of pigs (Jensen and Jorgensen, 1994). In total, however, CH_4 emission from enteric fermentation of pigs is only 0.2 million ton per year compared to 2.19 for dairy cattle and 2.31 for other cattle in western Europe (Steinfeld et al., 2006). Based on these numbers, we assumed that the difference in enteric CH_4 emission produced when barley instead of wheat middlings were used in diet of pigs only slightly affected emissions from *process B1a* and, therefore, *B1a* was set to zero.

ΔB1b is the difference in environmental impact of *process B1b*, i.e. feed utilization of a cow when feeding barley instead of wheat middlings. In dairy farming, CH_4 contributes approximately 52% to total GHG emissions in the chain, mostly caused by enteric fermentation processes within the cow (Gerber et al., 2010). As the amount of enteric CH_4 is related to the type and amount of feed (Dijkstra et al., 2007; Beauchemin et al., 2008; Ellis et al., 2008), we assumed a change in enteric CH_4 emission by dairy cattle due to feeding wheat middlings instead of barley (i.e. *B1b*). This change in enteric CH_4 emission can be computed by using IPCC Tier 2 or IPCC Tier 3 approach (IPCC, 2006). IPCC Tier 2 assumed that 6.5% of gross energy intake is converted to CH_4 (IPCC, 2006). As we exchanged beet tails and barley on the basis of their energy intake, no difference would be found. We, therefore, preferred IPCC Tier 3, which advices to use more specific data when possible. Based on empirical relationships between dry matter intake of different feed ingredients and CH_4 emission factors per ingredient, enteric CH_4 from dairy cattle was calculated. We adapted

CH_4 emission factors per feed ingredient from Vellinga et al. (2013), which are based on a mechanistic model originating from Dijkstra et al. (1992), and updated by Mills et al. (2001) and Bannink et al. (2006). For wheat middlings, enteric CH_4 emission was assumed to be 20.34 g/kg DM (DM content is 86.5%), and for barley 22.17 g/kg DM (Vellinga et al., 2013).

Situation 2: changing the application and the intermediate treatment

Situation 2, i.e. changing the application from *B1* to *B2* and the intermediate process of a co-product from *process I1* to *I2*, corresponds with the second case: beet tails fed to dairy cattle instead of using it as a substrate for anaerobic digestion. The environmental impact of this change in application and intermediate process is determined

$$D1-\Delta B1-I1+I2-D2+\Delta B2. \qquad (2)$$

Parameters are explained in the following paragraphs.

Computing *D1* and *D2*

D1 is the environmental impact related to the production of product D1 that is needed to replace product I1 in *process B1*. In our case of beet tails, multiple products are produced using the production of bio-energy and, therefore, D1 consists of two components: a and b (Figure 3). To quantify *D1*, we needed to determine the amount of marginal fossil-based electricity, heat (*D1a*), and artificial fertilizer (*D1b*) required to replace the bio-energy produced and fertilising capacity provided by 1 kg of beet tails and the resulting digestate. Accordingly, the environmental impact of producing these marginal products was included. Electricity and heat produced from bio-energy, and electricity and heat produced from fossil sources were exchanged on the basis of an equivalent amount of MJ. Digestate produced during the production of bio-energy and artificial fertilizer was exchanged on the basis of the N, P, and K fertilizer replacement value. The N fertilizer replacement value for digestate was assumed to be 65% and for artificial fertilizer 100% (DR 2012). Based on De Vries et al. (2012a), we assumed that the replacement value for P and K was 100% for all products. With 1 ton of beet tails, 459 MJ electricity, 240 MJ heat (i.e. 50% of the surplus heat produced), and 1.45 kg N (2.23×0.65), 0.70 kg P, and 2.30 kg K in digestate, were produced (De Vries et al., 2012b). The emissions and LU data for heat and electricity production from fossil energy and artificial fertilizer production were taken from the Ecoinvent database v2.2 (EcoinventCentre, 2007).

D2 is the environmental impact related to the production of product D2 that is avoided because of the use of product I2 in *process B2*. To quantify *D2* in our case of beet tails, we needed to determine the amount of barley avoided per kilogram of beet tails in dairy cattle feed, and the environmental impact and LU of barley (similar as in case 1). Just like for case 1, barley and beet tails were assumed to be exchanged in dairy cattle feed on the basis of their net energy content. Beet tails contain 106 VEM per kg, whereas barley contains 975 VEM per kg (CVB, 2010). To replace 1 kg of beet tails in diets of dairy cattle, we needed 0.11 kg of barley. Given a DM content of barley of 86.9%, 94 kg of DM barley was needed to replace 1 ton of beet tails (0.11 kg barley×0.869×1000=94).

Computing *ΔB1* and *ΔB2*

ΔB1 is the difference in environmental impact of *process B1*, i.e. application of fertilizer (*B1a*), electricity, and heat (*B1b*). We assumed that using electricity and heat produced from fossil sources instead of bio-energy will not have any impact. However, the difference in emissions during the application of artificial fertilizer instead of digestate has to be accounted for. Emission and LU of the application of 1 kg digestate and artificial fertilizer were based on data of EcoinventCentre (2007) and De Vries et al. (2012b).

ΔB2 is the difference in environmental impact of *process B2*, i.e. feed utilization of a cow. For dairy cattle, similar to the first example, the approach of Vellinga et al. (2013) was used. One kilogram beet tails DM (DM content is 13.6%) caused 20.00 g CH_4 emission (Vellinga et al., 2013).

Computing *I1* and *I2*

I1 is the environmental impact related to the production of product I1. Intermediate processes related to the production of bio-energy were transport of beet tails from the sugar factory to the bio-energy installation, digestion of beet tails, storage and transport of digestate, and burning of bio-energy.

I2 is the environmental impact related to the production of product I2. The only intermediate process was the transport of the beet tails from the sugar fabric to the dairy cattle farm.

Emissions and LU data were taken from the EcoinventCentre v2.2 (EcoinventCentre, 2007) and were based on De Vries et al. (2012b).

Situation 0: a co-product that currently goes to waste will be applied in a production process

Situation 0 is changing from *process W* to *process I*. Situation 0, however, only occasionally occurs in the livestock sector because most co-products used in animal feed already had an application. We, therefore, did not further elaborate on this situation with a case but only described the affected processes. The environmental impact of situation 0 is determined as:

$$I\text{-}W+\Delta B\text{-}D. \quad (3)$$

I is the environmental impact related to the production of product I, i.e. potato peels as feed ingredient. The volume of the intermediate treatment will increase, resulting in an increase of emissions from the intermediate process *(I)*. *W* is the environmental impact related to waste treatment. The volume of waste treatment will decrease *ΔB* is the change in environmental impact of *process B* due to using product I instead of product D. *D* is the environmental impact related to the production of product D. The volume of product D that fulfilled the application before product I will decrease.

3 Results and discussion

3.1 Results case study 1: wheat middlings fed to dairy cattle instead of pigs

Changes in land use

Changes in LU from using 1 ton wheat middlings in dairy cattle feed instead of in pig feed were based on Equation (1) : *D1a-ΔB1a-D1b+ΔB1b*= 996-0-1165+0= -169 m^2 (Figure 2).

Displacing 1 ton wheat middlings in pig feed required an additional production of 621 kg of DM barley, resulting in an increase of 996 m^2 (*D1a*). Using 1 ton wheat middlings as dairy cattle feed, however, displaced 726 kg of DM barley in dairy cattle feed, resulting in a decrease of 1165 m^2 (*D1b*). No LU was related to B. This means in our case study that land use was decreased with 169 m^2 when 1 ton of wheat middlings was used in dairy cattle feed instead of in pig feed.

Changes in emission of GHGs

Changes in emission of GHGs from using one ton wheat middlings in dairy cattle feed instead of in pig feed were based on Equation (1): *D1a-ΔB1a-D1b+ΔB1b*= 273-0-320+37= -9 kg CO_2-eq. (Figure 2).

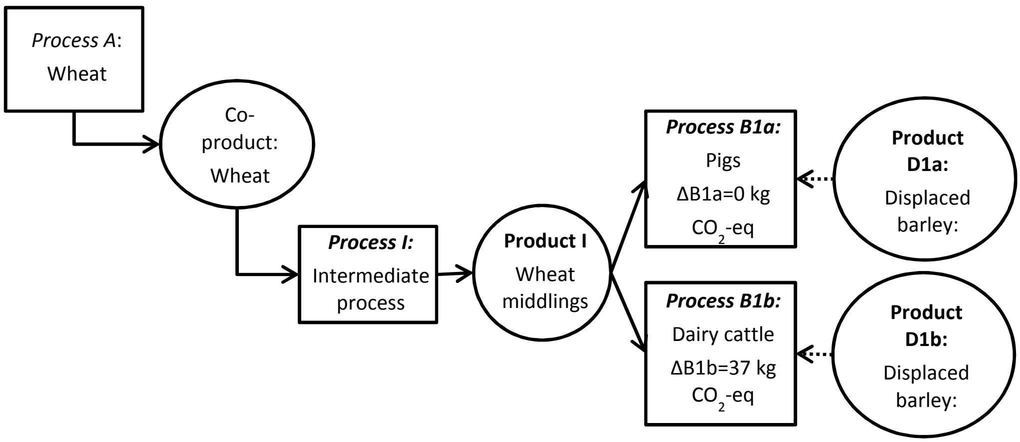

Figure 2. Processes that are affected by using one ton of wheat middlings as dairy cattle feed instead of using it as pig feed (Situation 1, Eq.: D1a-ΔB1a-D1b+ΔB1b).

Displacing 1 ton wheat middlings in pig feed required an additional production of 621 kg of DM barley resulting in an increase in GHG emission of 273 kg CO_2-eq (*D1a*). Using 1 ton wheat middlings as dairy cattle feed, however, displaced 726 kg DM barley in dairy cattle feed resulting in a decrease of 320 kg CO_2-eq (*D1b*).

GHG emission related to feed utilization (i.e. CH_4 from enteric fermentation) was 0 for pigs (*ΔB1a*) and 37 kg CO_2-eq for dairy cattle (*ΔB1b*). The latter value was computed given that 1 ton of wheat middlings as dairy cattle feed displaced 726 kg DM barley resulting in the emission of 16.10 kg CH_4 (i.e.726×22.17/1000), whereas 1 ton of wheat middlings (865 kg DM) resulted in the emission of 17.59 kg CH_4 (i.e. 865×20.34/1000). Therefore, enteric CH_4 emission was increased with 1.49 kg CH_4, resulting in 37 kg CO_2-eq by feeding wheat middlings instead of barley to dairy cattle.

This means, in our case study, that using 1 ton of wheat middlings in dairy cattle feed instead of in pig feed decreased GHG emissions by 9 kg CO_2-eq. When accounting for GHG emissions from LUC (i.e. 1.22×1.55×169 m^2 =−319 kg CO_2-eq), using 1 ton of wheat middlings in dairy cattle feed instead of in pig feed reduced emissions by 329 kg CO_2-eq. In this case, we assumed that the unused 169 m^2 was used to cultivate barley and, therefore, reduced the amount of forest and grass-land that was converted worldwide to support the increasing demand for barley. We could also argue that the unused 169 m^2 can be changed into grassland or forestland resulting in a different emission factor. It is, however, more valid to assume that the land remained in use of agricultural production and, therefore, does not result in C sequestration. It is, however, difficult to determine the consequences of a change in diet on LUC due to the complexity of the global feed market. Furthermore, it should be noted that Tonini et al. (2012) used an uncertainty of ±170 ton CO_2 ha^{-1}, resulting in a high uncertainty when LUC is incorporated in the results.

3.2 Results case study 2: use of beet tails in dairy cattle feed instead of using it as a substrate for anaerobic digestion

Changes in land use

Changes in LU from using one ton beet tails in dairy cattle feed instead of using it to produce bio-energy were based on Equation (2): $D1a,b$-$\Delta B1a,b$-$I1$+$I2$-$D2$+$\Delta B2$= 1.15-0.11-3.33+0.14-52+0= -154 m^2 (Figure 3).

Reducing one ton of beet tails to produce bio-energy resulted in an increase of LU from electricity of fossil sources of 0.79 m^2, 0.02 m^2 for heat, and 0.34 m^2 for artificial fertiliser. LU related to the transport of artificial fertiliser was <0.00 ($D1a,b$=0.79+0.02+0.34= 1.15 m^2). Using one ton beet tails as dairy cattle feed, however, displaced 94 kg DM barley in dairy cattle feed, resulting in a LU of 152 m^2 ($D2$). LU related to the intermediate process of digestion of beet tails ($I1$) was 0.14 m^2 for transport of beet tails, 2.81 m^2 for capital goods, 0.11 m^2 for digestion of beet tails, 0.27 m^2 for storage and transport of digestate, and no LU for burning of bio-energy ($I1$=0.14+2.81+0.11+0.27= 3.33 m^2). LU related to the intermediate process of beet tails fed to dairy cattle ($I2$) was the transport of beet tails, resulting in a LU of 0.14 m^2. LU related to the application ($\Delta B1a$) of the digestate was 0.13 m^2 instead of 0.02 m^2 for artificial fertiliser, resulting in a net LU of 0.11 m^2. ($\Delta B1a$=0.13-0.2= 0.11 m^2). No LU was related to $\Delta B1b$ and $\Delta B2$.

This means, in our case study, that land use was decreased with 154 m^2 when one ton of beet tails was used in dairy cattle feed instead of using it as a substrate for anaerobic digestion.

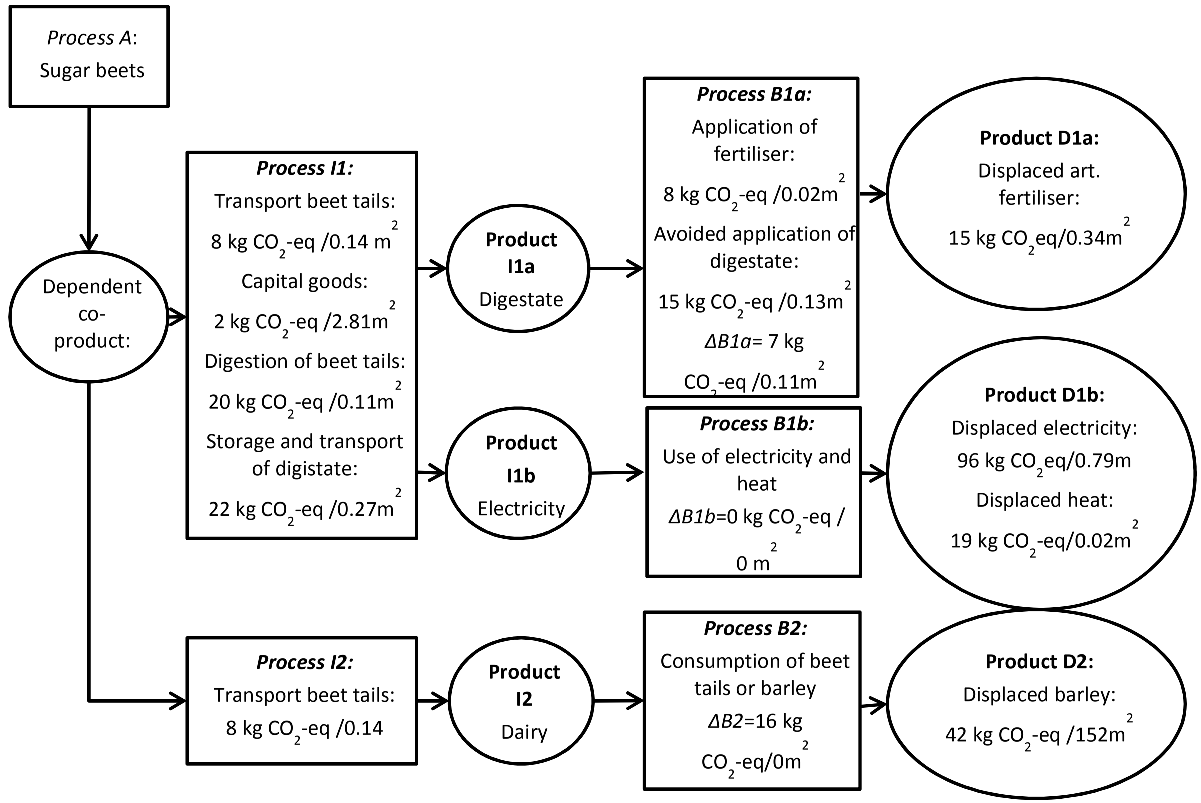

Figure 3. Processes that are affected by using one ton of beet tails as dairy cattle feed instead of using it as a substrate for anaerobic digestion (Situation 2, Eq.: D1a,b-ΔB1a,b-I1+I2-D2+ΔB2).

Changes in emission of GHGs

Changes in emission of GHGs from using 1 ton beet tails in dairy cattle feed instead of using it as a substrate for anaerobic digestion were based on Equation (2): D1a,b-ΔB1a,b-I1+I2-D2+ΔB2= 130-7-55+8-42+16= 50 kg CO_2-eq. (Figure 3).

Reducing 1 ton of beet tails to produce bio-energy resulted in an increase of 96 kg CO_2-eq from electricity, 19 kg CO_2-eq from heat, and 15 kg CO_2-eq from artificial fertilizer. Emissions related to the transport of artificial fertilizer were <0. (*D1a,b*=96+19+15=130 kg CO_2-eq). Using 1 ton beet tails as dairy feed, however, displaced 94 kg DM barley in dairy feed, resulting in a decrease of 42 CO_2-eq (*D2*).

GHG emissions related to the intermediate process of digestion of beet tails (*I2*) were 8 kg CO_2-eq for transport of beet tails, 2 kg CO_2-eq for capital goods, 20 kg CO_2-eq for digestion of beet tails, 22 kg CO_2-eq storage and transport of digestate, and 3 kg CO_2-eq for burning of biogas (*I1*=8+2+20+22+3=55 kg CO_2-eq). GHG emissions related to the intermediate process of beet tails fed to dairy cattle (*I2*) was the transport of beet tails, resulting in 8 kg CO_2-eq. GHG emission related to the application (*ΔB1a*) of digestate was 15 kg CO_2-eq, and the application of artificial fertilizer was 8 kg CO_2-eq, resulting in 7 kg CO_2-eq. GHG emissions related to feed utilization (i.e. CH_4 from enteric fermentation) was 16 kg CO_2-eq for dairy cattle (*ΔB2*). The latter value was computed given that 1 ton of beet tails in diets of dairy cattle displaced 94 kg DM barley resulting in emission of 2.09 kg CH_4 (i.e. 94×22.17/1000), whereas 1 ton of beet tail (136 kg DM) resulted in an emission of 2.72 kg CH_4 (i.e. 136×20/1000). Therefore, enteric CH_4 emission increased by 0.63 kg CH_4, resulting in 16 kg CO_2-eq by feeding beet tails instead of barley to dairy cattle.

This means, in our case study, that using 1 ton of beet tails in dairy cattle feed instead of using it as a substrate for anaerobic digestion, increased GHG emissions by 50 kg CO_2-eq. When accounting for GHG emissions from LUC (i.e. 1.22×1.55×−152 m^2 =−290 kg CO_2-eq), using 1 ton of beet tails in dairy cattle feed instead of producing bio-energy reduced emissions by 239 kg CO_2-eq. However, again, one should take into account an uncertainty of ±170 ton CO_2 ha^{-1} and, furthermore, we assumed again that the unused 154 m^2 was used to cultivate barley resulting in a reduction of converted forest or grassland.

4 General discussion

Nutritional requirements

In both cases, we assumed that we could replace the co-products wheat middlings and beat tails by one product (barley) based on the energy content. We based this assumption on the fact that wheat middlings, beet tails, and barley are all used for their energy content. Nevertheless, besides energy, there are other nutritional factors which should be taken into account, such as crude protein and amino acids. Furthermore, in the composition of diets, anti-nutritional factors and taste play a role. Increasing wheat middlings in diets of dairy cattle with 1 kg will most likely result in a decrease of multiple ingredients, including barley. An in-depth study is needed to analyse the nutritional consequences of changing the amount of co-products, such as wheat middling and beet tails, in diets of livestock. The same applies for the use of co-products to produce bio-energy. Beet tails are generally co-digested with manure and other substrates, but for reasons of simplicity, we focused solely on beet tails as substrate for anaerobic digestion.

Framework of Weidema et al. (2009) and our extended framework

Table 1. Equations for dealing with multifunctional processes, based on Weidema et al. (2009), p15.

Processes affected by a change in demand. for:	Product A	Product B
Dependent co-product fully utilised	A+I-D+ΔB	D+B

We based our framework on the theory of Weidema et al. (2009). They described their procedure for dealing with multifunctional processes on the basis of Table 1. They stated that if the demand for the determining product (product A) is increasing and if the co-product is fully used the processed co-product (product I) will replace product D. An increase in demand for product B, for which the co-product is used, will not result in an increase of *process A* because the production volume remains restricted to the production volume of the determining product (product A). The increased demand for product B, in this case, has to be supplied with product D.

We, however, wanted to assess the consequences of changing the application of a co-product. By doing so, we were able to analyse the optimal use of the co-products wheat middlings and beet tails. We, therefore, assumed that the demand of the determining product and product B remained equal.

General application of the framework

The framework provides assistance in how to assess the environmental impact of changing the application of a co-product. In this article, the focus was on increasing the use of co-products within the livestock sector. The theoretical framework, however, can be used also in sectors outside livestock production. For example, assessing the environmental impact of changing the application of wood shavings, a co-product produced during the production of laminate, from compost to bio-energy production. It should be noted that depending on the case, different impact categories can be used.

5 Conclusions

Based on an extended framework, we calculated the environmental consequences of using co-products in animal feed. We included environmental consequences for the alternative application of that used co-product and illustrated this by two cases: using wheat middlings as dairy cattle feed instead of pig feed and using beet tails as dairy cattle feed instead of a substrate for anaerobic digestion. Using wheat middlings in diets of dairy cattle instead of diets of pigs resulted in a decrease of 329 kg CO_2-eq per ton wheat middlings and a decrease of 169 m^2 land. Increasing the use of beet tails in diets of dairy cattle instead of using it as a substrate for anaerobic digestion resulted in a decrease of 239 kg CO_2-eq per ton beet tails and a decrease of 154 m^2 land. This indicates that increasing the use of wheat middlings and beet tails in diets of dairy cattle potentially can reduce GHG emissions and LU. However, emissions from LUC had a significant impact on the results. Excluding emissions from LUC in case 1 'wheat middlings' resulted in a decrease of 9 kg CO_2-eq and an increase of 50 kg CO_2-eq for case 2 'beet tails'. It should, however, be noted that (Tonini et al., 2012) used an uncertainty of ±170 ton CO_2 ha^{-1}, resulting in a high uncertainty when LUC is incorporated in the results.

Assessing the use of co-products in the livestock sector is of importance as the results of this study show that shifting the application of a co-product can reduce the environmental impact of the livestock sector. A correct assessment of the environmental consequences of using co-products in animal feed should also include potential changes in impacts outside the livestock sector, such as the impact in the bio-energy sector.

6 Acknowledgments

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Chapter 5

From environmental nuisance to environmental opportunity: housefly larvae convert waste to livestock feed

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Abstract

The livestock sector is in urgent need for more sustainable feed sources, because of the increased demand for animal-source food and the already high environmental costs associated with it. Recent developments indicate environmental benefits of rearing insects for livestock feed, suggesting that insect-based feed might become an important alternative feed source in the coming years. So far, however, this potential environmental benefit of waste-fed insects is unknown. This study, therefore, explores the environmental impact of using larvae of the common housefly grown on poultry manure and food-waste as livestock feed. Data were provided by a laboratory plant in the Netherlands aiming to design an industrial plant for rearing housefly larvae. Production of 1 ton dry matter of larvae meal directly resulted in a global warming potential of 770 kg CO_2-eq, an energy use of 9329 MJ and a land use of 32 m^2, caused by use of water, electricity, and feed for flies, eggs and larvae. Production of larvae meal, however, also has indirect environmental consequences. Food-waste, for example, was originally used for production of bio-energy. Accounting for these indirect consequences implies, e.g., including the environmental impact of production of energy needed to replace the original bio-energy function of food-waste. Assuming, furthermore, that 1 ton of larvae meal replaced 0.5 ton of fishmeal and 0.5 ton of soybean meal, the production of 1 ton larvae meal reduced land use (1713 m^2), but increased energy use (21342 MJ) and consequently global warming potential (1959 kg CO_2-eq). Results of this study will enhance a transparent societal and political debate about future options and limitations of larvae meal as livestock feed. Results of the indirect environmental impact, however, are situation specific, e.g. in this study food-waste was used for anaerobic digestion. In case food-waste would have been used for, e.g., composting, the energy use and related emission of greenhouse gases might decrease. Furthermore, the industrial process to acquire housefly larvae meal is still advancing, which also offers potential to reduce energy use and related emissions. Eventually, land scarcity will increase further, whereas opportunities exist to reduce energy use by, e.g., technical innovations or an increased use of solar- or wind energy. Larvae meal production, therefore, has potential to reduce the environmental impact of the livestock sector.

1 Introduction

The livestock sector is in urgent need for alternative, more sustainable feed sources, because of the increased demand for animal-source food and the already high environmental costs associated with production of livestock feed. The current livestock sector is responsible for about 15% of the anthropogenic emissions of greenhouse gases (Gerber et al., 2013), mostly related to production and utilization of feed (De Vries and De Boer, 2010). The sector also increasingly competes for scarce resources, such as land, water, and fossil energy (Steinfeld et al., 2006; Godfray et al., 2010). Livestock production currently uses about 70% of the agricultural land (Steinfeld et al., 2006), mainly for pasture and production of feed crops. Expansion of the area for livestock production leads to deforestation in the tropics, i.e. 80% of new croplands are replacing forest, resulting in biodiversity loss and increased carbon emissions (Foley et al., 2007; Gibbs et al., 2010; Foley et al., 2011). Without major changes, therefore, the above described environmental concerns will only increase further. One of the major challenges, therefore, is sustainable production of livestock feed.

Recent developments indicate environmental benefits of rearing insects for livestock feed (Van Huis et al., 2013; Sánchez-Muros et al., 2014). Insects have a low feed conversion ratio (kg dry matter feed/kg product) and can be consumed completely, without residual materials as bones or feathers. The nutritional value of insects is high, especially as a protein source for livestock (Veldkamp et al., 2012). Insect-based feed products, therefore, can replace conventional feed ingredients, like fishmeal or soybean meal (SBM), which are associated with a high environmental impact (Veldkamp et al., 2012; Van Huis et al., 2013). The use of insects may reduce the environmental impact of livestock production. In contrast with cultivation of feed crops, production of insects is not necessarily land intensive, especially because insects can turn organic waste streams, such as manure or food-waste, into high-quality insect-based feed products (Veldkamp et al., 2012; Van Huis et al., 2013; Sánchez-Muros et al., 2014). In Western countries large amounts of manure are produced and, according to the FAO one third of the produced food is never consumed (Gustavsson et al., 2011). Already in the 1970s, it was proven that housefly larvae (Musca domestica L.) can be used for biodegradation of chicken manure (Calvert et al., 1970) and that larvae can grow on municipal organic waste (Ocio et al., 1979). Moreover, feeding houseflies reared on manure and food-waste to livestock will reduce the competition for land between food and feed, because they can replace other feed ingredients that are directly edible by humans. As an example, about 70% of the cereal grains used in developed countries is fed to livestock (Eisler et al., 2014). Due to a rather inefficient feed conversion ratio of livestock - for chicken 1.6, for pigs 2.5 and cattle 5.1 (Šebek and Temme, 2009) - more people could be supported from the same amount of land if they did not consume meat from livestock fed with cereals (Godfray et al., 2010). Feeding waste-fed insects to livestock, therefore, might be an effective strategy as inedible waste streams for livestock and humans can be used to produce high-quality food products, such as meat, milk, and eggs.

Altogether, waste-fed insects seem to be a promising feed source for livestock and, therefore, can be part of the solution to fulfil the growing demand for animal-source food, within the carrying capacity of the earth.

To our knowledge, however, no study has been published that quantified the reduction of the environmental impact of including waste-fed insects in livestock feed. Only one peer-reviewed study analysed the environmental impact of insects, in this case meal-worms (Oonincx and De Boer, 2012). This study, however, focused on production of mealworms for human consumption, and showed that the production of one kg of edible protein from mealworms resulted in a lower land use (LU), but a higher energy use (EU), and consequently also a higher global warming potential (GWP) than production of one kg of edible protein from livestock (Oonincx and De Boer, 2012). It is questionable, therefore, whether or not the production of waste-fed insects will result in environmental benefits.

The aim of this study, therefore, is to explore whether the environmental impact of livestock production can be reduced by the use of larvae of the common housefly grown on organic waste streams as livestock feed.

2 Materials and methods

Life cycle assessment (LCA) was used to assess the environmental impact of larvae meal production. LCA is an internationally accepted and standardized holistic method (ISO 14044, 1997; ISO 14040, 2006) to evaluate the environmental impact during the entire production chain (Guinée et al., 2002; Bauman and Tillman, 2004). LCA includes four phases: goal and scope definition, inventory analysis (data collection), impact assessment (encompasses classification and characterization of the emissions and resources used), and interpretation of results.

Goal and scope definition

The goal of this study was to assess the environmental impact of livestock production when larvae of the common housefly grown on organic waste streams are used as livestock feed, including also the environmental consequences to replace the original application of this waste. The functional unit was 1 ton larvae meal on dry matter (DM) basis.

Inventory analysis

Data related to the required inputs and outputs to produce 1 ton of larvae meal were obtained from a business model. This model was based on experimental studies and developed by four companies in the Netherlands (Jagran, an insect rearing company, supported by AEB and SITA, two waste processing companies, and Denkavit, an animal nutrition company).

Impact assessment

To assess the environmental impact, two types of impacts were considered: use of resources, such as land or fossil-energy, and emission of pollutants, such as carbon dioxide or nitrous oxide (Guinée et al., 2002). The following impact categories were assessed: climate change, generally expressed as GWP, EU, and LU. Climate change and LU were chosen because the livestock sector contributes significantly to both emission of greenhouse gases and LU worldwide (Steinfeld et al., 2006). EU was included also because Oonincx and De Boer (2012) showed that rearing insects is energy demanding, and because fossil-energy is a scarce resource. The following greenhouse gases were considered: CO_2, CH_4, and N_2O. These greenhouse gases were summed based on their equivalence factors in terms of CO_2-eq (100 years' time horizon): i.e. 1 for CO_2, 25 for CH_4, and 298 for N_2O (Forster et al., 2007), and expressed per ton larvae meal (DM). LU was expressed in m^2 per ton larvae meal (DM) per year, whereas EU was expressed in MJ per ton larvae meal (DM). Data related to emissions and resources were mainly obtained from databases and literature and are described in more detail in the next paragraphs. In case of a multifunctional process (e.g. production of soybean oil and meal), economic allocation was used, which is the partitioning of environmental impacts between co-products based on the relative economic value of the outputs (Guinée et al., 2002). Economic allocation is most commonly used in LCA studies of livestock products (De Vries and De Boer, 2010).

The direct and indirect environmental impacts related to the production of 1 ton of larvae meal were assessed (Figure 1). Direct environmental impacts[1] resulted from the use of resources and emissions of pollutants related to the housefly farm, such as use of water, electricity or feed, and emissions of greenhouse gases from waste during insect rearing. The indirect environmental impacts[2] related to changes in use of farm inputs or outputs produced. Food-waste, for example, used for insect rearing, might have been used originally to produce bio-energy. To evaluate the impact of using food-waste for insect rearing, therefore, the environmental impacts of, for example, production of fossil-energy needed to

[1] Also referred to as attributional LCA in this thesis.

[2] Also referred to as consequential LCA in this thesis. The consequential LCA method used is based on the principles of consequential LCA framework developed in Chapter 4.

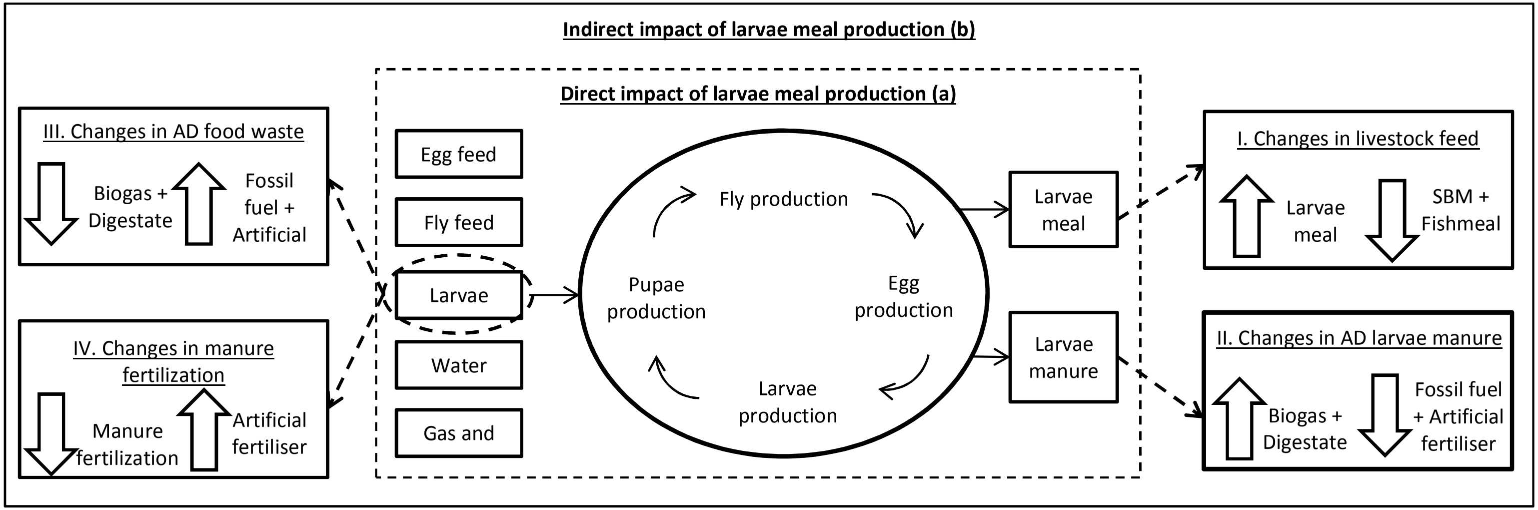

Figure 1. Production chain of larvae meal. Box (a) shows the life cycle of housefly larvae with the related input products that are required to maintain a certain output of larvae meal and larvae manure. Box (b) shows the four indirect environmental changes that will occur when larvae meal is used as livestock feed: changes in livestock feed, changes in anaerobic digestion (AD) and changes in manure fertilization.

replace the original bio-energy function of food-waste was included also. Below the direct environmental impact and indirect environmental impact are explained in more detail.

2.1 Direct environmental impact of larvae meal

Data were needed on all farm inputs and outputs related to the production of 1 ton larvae meal (DM) to assess the direct environmental impacts of housefly rearing. These data were based on a business model aimed at producing 65 ton of fresh larvae per day, resulting in 20 ton of larvae meal with a DM content of 88%. The production cycle started with housefly pupae, which eclosed within 2 days. Feed for the flies consisted of sugar, milk powder, and egg powder. Flies were kept at a temperature of 25° C. Female flies start to lay eggs after 7 days in an oviposition substrate, consisting of milk powder, yeast, fibre, vegetable oil, and premix (vitamins and minerals). After oviposition, the mixture of eggs with substrate was added to a larvae-substrate (feed for the larvae). The larvae-substrate consisted of food-waste, laying hen manure, and premix. Larvae were kept at a temperature of 27° C and were fully grown after 5 days. Per 4 kg of larvae-substrate, 1 kg of fresh larvae was produced. Harvesting of the larvae was performed by shutting off the ventilation, which forces the larvae to migrate to the surface of the substrate because of a drop in oxygen level. The harvested larvae were dried, which is generally required before the larvae can be included into livestock feed. As the larvae production was situated next to a waste incineration facility, the remaining heat of this facility was aimed for drying. After approximately 35 days the fly colony dies. Therefore, part of the larvae is kept to evolve into pupae to maintain the production chain.

Table 1. data and related global warming potential (GWP), energy use (EU) and land use (LU) data with references for the direct environmental impact of producing 1 ton dry matter larvae meal.

Ingredients	Unit	Amount (/ton DM)	GWP (g CO_2-eq)	EU (MJ)	LU (m^2)	References
Feed flies	kg	1	3808	12.2	1.34	Vellinga et al., 2013
Substrate eggs	kg	17	1351	3.9	0.34	Vellinga et al., 2013
Food-waste	kg	11079	11	0.2	0.00	Coenen et al., 2013; IPCC 2006
Manure	kg	3693	42	0.2	0.00	EcoinventCentre, 2007; IPCC 2006
Premix	kg	57	1362	3.9	0.34	EcoinventCentre, 2007; IPCC 2006 Mosnier et al., 2011
Water	kg	10309	0	0.0	0.00	EcoinventCentre, 2007
Electricity	kWh	378	753	11.8	0.01	EcoinventCentre, 2007
Gas	kWh	183	586	11.2	0.00	EcoinventCentre, 2007

A summary of the environmental impacts of input and output data required to maintain the production of larvae meal is provided in Table 1 and further explained below.

The environmental impacts (i.e. GWP, EU and LU) related to the production of feed for the flies, and egg substrate were based on Mosnier et al. (2011) and Vellinga et al. (2013) and exact composition data, which are not disclosed for industrial competitive protection. Environmental impacts from production of various feed ingredients included impacts from cultivation (e.g. fertilizers, pesticides, machinery, energy, emissions related to direct and in-direct N_2O, and CO_2 emissions from liming and urea fertilization), impacts from drying and processing, and impacts from transport up to the farm gate.

The environmental impacts (i.e. GWP, EU, and LU) related production and use of tap water, natural gas, and electricity were based on Eco-invent (EcoinventCentre, 2007). Electricity was assumed to be substituted with marginal Dutch electricity, i.e. 28% coal-based, 67% natural gas-based, and 5% wind-based electricity (De Vries et al., 2012a).

Emissions of GHG related to the substrate for the larvae were based on IPCC (IPCC, 2006). According to IPCC, emissions of CH_4 from organic waste occur only after several months. As food-waste was used during the larvae production process for 4 days only, it was assumed that emissions from organic waste were negligible. During the handling and storage of laying hen manure, CH_4 and direct and indirect N_2O were emitted. As there were no specific data available about the use of manure for insect rearing, it was assumed that emissions for using manure were equal to emissions emitted on a laying hen farm. To estimate CH_4 emission, a tier 2 approach was used based on country specific data (Coenen et al., 2013) and IPCC default values (IPCC, 2006) (an organic matter content of 0.35 kg per kg manure, maximum CH_4 producing potential of 0.34 m^3 CH_4 per kg organic matter and a methane conversion factor of 0.015). To estimate direct and indirect N_2O emissions, a tier 2 approach was used based on country specific data of Coenen et al. (2013) (direct: 0.8 kg N excretion per laying hen per year, 18.9 kg manure per laying hen per year and a default emission factor of 0.01) (indirect: volatilization 40% and an emission factor of 0.01). The GWP, EU, and LU for transportation of food-waste and manure over an average of 65 km per day were included, based on Ecoinvent (EcoinventCentre, 2007).

2.2 Indirect environmental impact of larvae meal production

Three major indirect changes were considered. *I) Changes related to a decreased availability of food-waste for anaerobic digestion.* Originally, food-waste is used for anaerobic digestion in the Netherlands. As the amount of food-waste is restricted by the amount of food spilled by humans, using food-waste for larvae production decreases its availability for anaerobic digestion. The decreased production of electricity, heat and digestate, therefore, was assumed to be substituted with fossil-energy and synthetic fertilizer. *II) Changes related to manure fertilization.* Laying hen manure used to grow insects was assumed to be used to fertilize crop production in Germany. As the production of laying hen manure is restricted by the demand for eggs, its availability for application on croplands in Germany decreased. Consequentially, laying hen manure was assumed to be substituted with synthetic fertilizer. *III) Changes related to an increased availability of larvae manure for anaerobic digestion.* Larvae manure produced was assumed to be used originally for anaerobic digestion, producing electricity, heat, and digestate. Electricity and heat were assumed to replace fossil-energy. Digestate was assumed to replace synthetic fertilizer. As digestate contains residues of laying hen manure it should be labelled as manure according to Dutch regulation. Due to a manure surplus in the Netherlands, digestate was assumed to be transported to Germany, and used to fertilize crop production.

The method and related data to the above-described indirect changes in anaerobic digestion and manure fertilization are described below.

Environmental impact related to changes in anaerobic digestion

For the anaerobic digestion of food-waste and larvae manure a large scale digestion plant was considered. The biogas produced was used in a combined heat and power unit for the production of electricity and heat. Digestion required 110 MJ of electricity per ton and 65 MJ heat per ton based on 10% DM content (Berglund and Börjesson, 2006). Electricity was taken from the grid, whereas heat originated from the combined heat and power unit. Methane losses were 1% of produced CH_4 (Zwart et al., 2006; Møller et al., 2009). The energy efficiency of the heat and power unit was 70%, the electric efficiency 35% (Zwart et al., 2006). The utilization of surplus heat from anaerobic digestion, i.e. the surplus heat that remains after using the required heat for the process, was not included, as heat offset possibilities are limited in the Netherlands (Dumont, 2010). The digestate that is transported and applied to the field as fertilizer, was assumed to substitute marginal mineral N, P, and K fertilizer. Marginal production of mineral fertilizer was assumed to be calcium ammonium nitrate for N, triple superphosphate for P_2O_5, and potassium chloride for K_2O (De Vries et al., 2012a).

Per ton larvae manure (DM) 15 GJ was produced (Table 2). N, P, and K values and the methane production potential of larvae manure were provided by Jagran. Jagran performed two analyses of the methane production potential of larvae manure. The first analysis of the larvae manure was based on a sample in which the larvae grew only on food-waste and the second analysis was based on a sample in which the larvae grew only on laying hen manure. Results of both analyses were used based on their ratio in the diet (25% manure and 75% food-waste).

Per ton food-waste (DM) 18 GJ could be produced. No analysis was performed on the methane production potential and N, P, and K values of food-waste. It was, therefore, assumed that values for food-waste were similar to larvae manure based on a substrate of food-waste, when compensated for the difference in DM content. Both food-waste and larvae manure were transported to the digestion plant and the digestate to agricultural fields. It was assumed that digestate of food-waste was applied on Dutch agricultural fields and digestate of larvae manure on German croplands, as it contains laying hen manure and, therefore, is not allowed on Dutch fields, resulting in a difference in transport of 370 km. The environmental impact values related to the production of electricity, and N, P, and K fertilizers and trans-port per lorry were based on Eco-invent (EcoinventCentre, 2007) (Table 3).

Environmental impact related to changes in manure fertilization

Laying hen manure was transported to Germany (435 km assumed) and used as fertilizer replacing artificial fertilizer. Laying hen manure contains 3.57% N, 2.89% P, and 2.03% K (Den Boer et al., 2012).

Table 2. Input data anaerobic digestion per ton of organic matter on dry matter basis.

Type	DM	biogas	CH_4 in biogas	Energy	N	P	K
		nm^3/ton	%	GJ/ton DM	%	%	%
Larvae manure: food-waste	38	260	65	17	3.28	0.76	0.98
Larvae manure: laying hen manure	30	103	56	8	2.86	3.32	2.99
Food-waste	32	218	65	18	2.76	0.64	0.50

Table 3. Global warming potential (GWP), energy use (EU), and land use (LU) data with references for the indirect environmental impact of producing 1 ton dry matter larvae meal.

Ingredients	Unit	GWP	EU	LU	References
	(/ton DM)	(g CO_2-eq)	(MJ)	(m^2)	
Soybean meal	kg DM	710	6.89	3.543	Vellinga et al., 2013
Soybean meal incl LUC	kg DM	2375	6.89	3.543	De Vries et al., 2012b
Fishmeal	kg DM	1636	23.45	0.015	Vellinga et al., 2013
Electricity	kWh	753	11.80	0.006	EcoinventCentre, 2007
N	kg	8543	55.35	0.095	EcoinventCentre, 2007
P_2O_5	kg	2014	25.96	0.087	EcoinventCentre, 2007
K_2O	kg	495	8.06	0.051	EcoinventCentre, 2007
Lorry EURO 4 16-32 ton	Tkm	164	2.57	0.003	EcoinventCentre, 2007

2.3 Comparison of environmental impact of larvae meal with SBM and fish meal

To determine the potential environmental benefit of using larvae meal as livestock feed ingredient, all above described indirect environmental impacts were added to the direct environmental impacts of production of 1 ton larvae meal (DM). Subsequently, the environmental impact of larvae meal was compared with other protein rich feed ingredients, namely SBM and fishmeal. One ton of larvae meal was assumed to replace 0.5 ton of fishmeal and 0.5 ton of SBM on a DM basis. The DM content of larvae meal is 88.0% (based on analysis of the laboratory plant), of fishmeal 92.7% and of SBM 87.5% (CVB, 2012). Table 3 shows the GWP, EU, and LU for fishmeal and SBM. For SBM emissions related to LUC were assessed as well, because the production of SBM is related to deforestation which is an important source of GHG emissions (Prudêncio da Silva et al., 2010; Macedo et al., 2012; Van Middelaar et al., 2013) LUC emissions for SBM were 0.47 kg CO_2-eq per m^2 per year, assuming an amortization period of 20 years (De Vries et al., 2012b). Emissions were quantified by considering CO_2 emissions of converting, for example, forest or grassland to cropland, accounting for size and location of converted land and the types of land that were converted (biome types) (Tonini et al., 2012). It was assumed that 20% of the increased soybean demand came from increased yields, whereas 80% was met by expansion of land in Brazil (Kløverpris 2008; Laborde, 2011) of which 23% rainforest and 77% savannah in the Cerrado region (Prudêncio da Silva et al., 2010). For larvae meal and fishmeal it was assumed that LUC emissions were negligible, because land use during production is small.

5

2.4 Sensitivity analysis

Inventory data and data related to emission of GHGs, EU, and LU contain uncertainties. A sensitivity analysis was performed in Excel to assess which parameters contained high uncertainty and, therefore, have a high impact on the outcome of the study. Inventory data regarding, for example, the amount of electricity or water used, were changed one by one by ±10%[3]. Moreover, the environmental impact (GWP, EU, and LU) related to production of these products was also changed by ±10%. This range of ±10% is more often applied in LCA studies of livestock products, such as in Van Middelaar et al. (2013).

[3] Also referred to as one-at-the-time approach (OAT) in this thesis.

3 Results

First the results of the direct environmental impacts are shown, followed by the results of the indirect environmental impacts. Then a comparison of larvae meal with SBM and fish meal is made, and lastly results of the sensitivity analysis are presented.

3.1 Direct environmental impact of production of larvae meal

Producing larvae meal resulted in a GWP of 770 kg CO_2-eq, an EU of 9329 MJ, and an LU of 32 m^2 per ton DM larvae meal (Table 4). The largest part of the GWP was caused by feed for the larvae (44%), whereas an additional 37% resulted from the use of electricity and 14% form the use of gas. Electricity and gas use, however, explained the majority of the EU (70%), whereas production of vitamins and minerals in larvae feed explained the majority of the LU. Note that gas for drying the larvae, which is currently assumed to be necessary before it can be used as livestock feed due to food safety issues, was not included as the remaining heat from a waste incineration facility was used. However, when no residual heat could have been used, one should count for an additional 1247 kg CO_2-eq, 23949 MJ and 1 m^2 per ton larvae meal (DM). Therefore, gas for drying will have a large impact on EU and consequently on the GWP.

Table 4. Global warming potential (GWP), energy use (EU), and land use (LU) of the production of 1 ton of larvae meal dry matter.

Processes		**GWP** (kg CO_2-eq)	**EU** (MJ)	**LU** (m^2)
Egg production		*26*	*84*	*7*
	Egg substrate	23	67	6
	Feed flies	2	7	1
	Water	1	11	0
Larvae production		*353*	*2733*	*23*
	Feed larvae	350	2686	22
	Water	3	46	0
Electricity use		*284*	*4458*	*2*
	Egg production	21	322	0
	Larvae production	164	2575	1
	Processing larvae	41	644	0
	Lightening building	38	595	0
	Working places	21	322	0
Gas for heating		*107*	*2054*	*0*
Total larvae meal		*770*	*9329*	*32*

3.2 Indirect environmental impacts related to changes in farm inputs and outputs

Impacts related to changes in anaerobic digestion of food-waste

The increased use of fossil-energy, due to the reduced anaerobic digestion of food-waste, increased GWP with 3954 kg CO_2-eq, EU with 62001 MJ, and LU with 32 m^2 per ton larvae meal (DM), whereas the increased use of synthetic fertilizer increased GWP with 895 kg CO_2-eq, EU with 6230 MJ, and LU with 13 m^2 per ton DM of larvae meal (Figure 2).

Impacts related to changes in anaerobic digestion of larvae manure

The decreased use of fossil-energy, due to the increased use of anaerobic digestion of larvae manure, reduced GWP with 3277 kg CO_2-eq, EU with 50916 MJ, and LU with 27 m^2 per ton DM larvae meal (Figure 2). The decreased use of synthetic fertilizer reduced GWP with 355 kg CO_2-eq, and LU with 5 m^2, but increased EU with 2825 MJ per ton larvae meal (DM), due to transport of larvae manure to Germany (Figure 2).

5

Impacts related to changes in manure fertilization

The increased use of synthetic fertilizer, due to the reduced use of manure fertilization, increased GWP with 1146 kg CO_2-eq, EU with 7045 MJ, and LU with 22 m^2 per ton DM of larvae meal (Figure 2).

3.3 Comparison of environmental impact of larvae meal with SBM and fish meal

To determine the potential environmental benefit of larvae meal, the direct environmental impact was added to the indirect environmental impacts of producing 1 ton larvae meal (DM). Production of 1 ton larvae meal (DM) resulted in a GWP of 3132 CO_2-eq, an EU of 36513 MJ and an LU of 66 m^2. Subsequently, the environmental impact of 1 ton larvae meal was compared with fishmeal and SBM. Using larvae meal instead of SBM and fishmeal resulted in an increased GWP of 1959 kg CO_2-eq (i.e. excluding LUC) or 1364 kg CO_2-eq (i.e. including LUC), EU with 21342 MJ, and LU decreased with 1713 m^2 per ton larvae meal (DM).

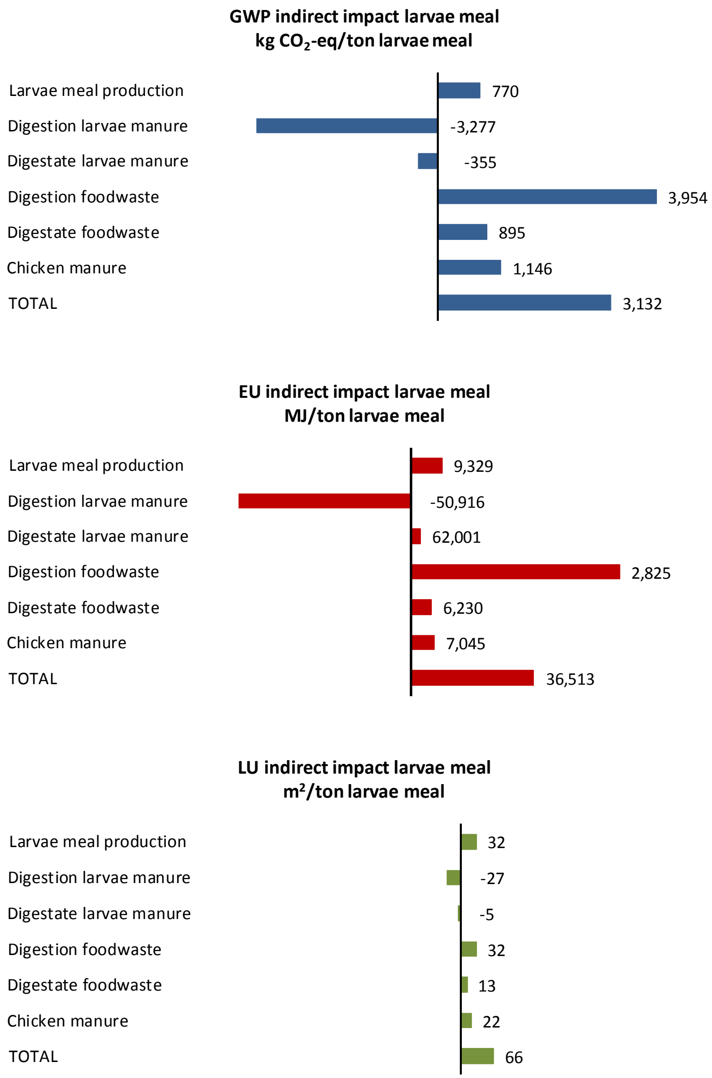

Figure 2. Indirect environmental impact of larvae meal production. Global warming potential (GWP), energy use (EU), and land use (LU) per kg of dry matter larvae meal for indirect changes of producing larvae meal. By summing up the environmental changes the total indirect environmental impact of larvae meal is obtained.

3.4 Sensitivity analysis

Results of the sensitivity analyses showed that the direct GWP was merely determined by energy use (gas and electricity use) and feed for larvae (Table 5), whereas EU was merely determined by electricity use of larvae production, followed by gas use for the total building. LU was mostly influenced by the land used for producing the feed of the larvae.

The indirect GWP and EU however, were merely determined by changes in anaerobic digestion (Table 6). This sensitivity for the process of anaerobic digestion has two causes. First, EU and GWP outcomes highly depend on the methane production potential influencing the amount of energy assumed to be produced by anaerobic digestion. Second, EU and GWP highly depend on the electricity factor used for greenhouse gas emissions, which was merely determined by the mixer of electricity sources (in this cased based on the Dutch situation). LU was mostly influenced by the land used for the production of SBM, and LU outcomes, therefore, were sensitive to changes in the relative replacement of SBM and fishmeal by larvae meal.

Table 5. Sensitivity analysis of direct impact of larvae meal production. Consequences of 10% change in emission factors on global warming potential GWP, energy use (EU), and land use LU of larvae meal per ton larvae meal.

Direct changes		**GWP** (kg CO_2-eq)	**EU** (MJ)	**LU** (m^2)
Larvae meal (total)		770	9329	32
Egg production		*3*	*8*	*1*
	Egg substrate	2	7	1
	Feed flies	0	1	0
	Water	0	1	0
Larvae production		*35*	*273*	*2*
	Feed larvae	35	269	2
	Water	0	5	0
Electricity use		*28*	*446*	*0*
	Egg production	2	32	0
	Larvae production	16	257	0
	Processing larvae	4	64	0
	Lightening building	4	60	0
	Working places	2	32	0
Gas for heating		*11*	*205*	*0*

Table 6. Sensitivity analysis of indirect impact of larvae meal production. Consequences of 10% change in impact factors and inventory data on global warming potential (GWP), energy use (EU), and land use (LU) of 1 ton larvae meal dry matter.

Indirect changes		**GWP** (kg CO_2-eq)	**EU** (MJ)	**LU** (m^2)
Larvae meal production (incl. indirect change)		1959	21342	-1713
Livestock feed				
	Impact factor larvae meal	77	933	3
	Impact factor fishmeal	81	345	1
	Impact factor soybean meal	35	1173	177
	Ratio fishmeal and soybean meal	92	1656	353
Anaerobic digestion larvae manure				
	Methane production potential	374	5801	3
	Impact factor electricity	325	5092	3
	EF N,P,K	100	726	2
	% N,P,K	100	726	2
Anaerobic digestion food-waste				
	Methane production potential	457	7152	4
	Impact factor electricity	407	6385	3
	Impact factor N,P,K	89	623	1
	% N,P,K	89	623	1
Laying hen manure				
	% N,P,K	138	1067	2

4 Discussion

Results of the sensitivity analysis showed that the following four input data are highly sensitive: electricity and gas use, emissions related to the feed of the larvae, replacement of livestock feed, and the methane production potential. Those parameters are below discussed in more detail.

Electricity and gas use were the main contributors to EU and the GWP during the production of larvae meal. A relative high electricity use and gas use for insect rearing were found earlier for mealworm production (Oonincx and De Boer, 2012). Production of one kg of mealworm (DM) used 15.8 MJ of electricity and 26.0 MJ of gas. The higher energy use for production of mealworms was caused by a longer production cycle of mealworms (10 weeks instead of 5 days). Production of mealworms and housefly larvae use high amounts of gas due to the required ambient temperature of the insects. One should, however, take into account that the required energy use was an estimation of the laboratory plant. The industrial process to acquire housefly larvae meal is still advancing, which offers potential to further reduce energy use. Taelman et al. (2013) showed, for example, that up-scaling production of algae as feed for aquaculture reduced the carbon footprint with a factor 20. It is likely, therefore, that further up-scaling of production of larvae meal will further reduce energy use and related GHG emissions.

Emissions of GHG related to larvae-feed contributed most to GWP. Calculations of the larvae feed were based on IPCC guidelines for manure and composting of food-waste. It was assumed in this study that emissions of food-waste were negligible. Despite food-waste is used by larvae for 4 days only, emissions of GHG could have occurred because the circumstances for composting were favourable (i.e. high temperatures and constant ploughing by the larvae). Furthermore, IPCC calculations for manure were based on emissions related to the complete laying hen sector and not only for the storage of manure and, therefore, possibly resulting in an over-estimation. To minimize the uncertainty of the environmental impact of larvae meal experimental studies on the emissions related to use of larvae feed are required.

The changes in livestock feed were based on the assumption that larvae meal is used to replace fishmeal and SBM. Although larvae meal, fishmeal and SBM are all protein rich, their nutritional value, i.e. content of amino acids and crude fat, differs. Table 7 shows the nutrient content of larvae meal (based on analysis of the laboratory plant), fishmeal and SBM (CVB, 2010). The nutrient content of larvae meal was based on two samples only, but results are confirmed by a literature review (Veldkamp et al., 2012), showing similar outcomes: larvae Contain 43-68% protein and 4-32% fat on a DM basis. Replacing fishmeal and SBM by larvae meal on a DM basis will have an impact on the nutritional composition of the diets such as crude protein or amino acids and net energy. Since the nutritional requirements should be met to maintain the growth performance of the animal, the diet composition will change. It is, therefore, expected that including larvae meal in livestock diets will not only reduce the content of fishmeal and/or SBM, but will affect the complete diet composition. Future research, therefore, should include changes in diet composition and changes on feed intake

Table 7. Nutrient content (%) of larvae meal (based on data of the laboratory plant), fishmeal and soybean meal (SBM) (CVB, 2010).

	Larvae meal	Fishmeal	SBM
Dry matter	88.0	92.7	87.5
Crude protein	47.9	56.7	46.0
Fat	24.2	15.8	18.4
Lysine	32.6	43.1	28.5
Methionine	11.3	15.9	6.4

and growth of the animals. However, until now the nutritional value of larvae meal is highly uncertain. In vivo animal studies are required to determine the palatability, digestibility and other relevant characteristics of larvae meal before a comparison with other feed ingredients can be made.

Assumed values of the methane production potential highly influenced the computation of the amount of energy produced from anaerobic digestion. The methane production potential used in this study, i.e. 15 GJ per ton larvae manure (DM) and 18 GJ per ton food-waste (DM), were within the range of values found in other studies: 12 GJ per ton DM of municipal organic waste (Berglund and Börjesson, 2006) and 18 GJ per ton DM of vegetable food-waste (Bernstad and la Cour Jansen, 2012). Nevertheless, the difference between the methane production potential of food-waste and larvae manure remains uncertain. To minimize the uncertainty of the environmental impact of larvae meal more experimental studies in this direction are required.

Although data are based on a Dutch case study (Jagran), the results are valuable for livestock systems across the globe. A global interpretation of current results, however, requires consideration of some site-specific aspects. First, a large part of the direct environmental impact is caused by energy use. Situating the production in a warmer climate than the Netherlands will lower gas requirements and, therefore, lower the GWP and EU. Second, results showed that it is essential to minimize gas use for drying to reduce the GWP and EU. Therefore, larvae production should be situated near, for example, a waste incineration facility, which enables to use its remaining heat for drying. Third, this study shows that the environmental impact of using larvae meal as livestock feed also depends on the current application of the food-waste. Using food-waste and laying hen manure as feed for larvae in this case study resulted in an increased GWP and EU but a decreased LU compared with the current situation, in which food-waste was used for anaerobic digestion and laying hen manure for fertilization. In case food-waste is used for composting or does not have a function yet, environmental benefits related to larvae meal will increase. The same applies for the application of the larvae manure. Finding the optimal application for larvae manure is essential to reduce the environmental impact. Finally, some challenges need to be addressed in Europe before large scale larvae production can start. Legislation, health concerns, and a high cost price are important issues (Veldkamp et al., 2012; Van Huis et al., 2013). Until now, it is not allowed to include larvae fed on waste streams in livestock feed in the Netherlands. However, in other parts of the world this is less strictly regulated.

The discussion points mentioned in this section provide building blocks to minimize the environmental impact of larvae production. Incorporating the results of this study within designing plans of large scale larvae production provide opportunities to lower GWP and EU while a low LU is maintained.

5 Conclusion

This study is the first study that explores the environmental impact of using larvae meal as livestock feed. Larvae meal can be a promising protein source for the livestock sector that can alleviate future shortages of protein sources in livestock feed. Results of this study will enhance a transparent societal and political debate about future options and limitations of larvae meal as livestock feed.

Production of 1 ton of dry matter larvae meal directly resulted in a global warming potential of 770 kg CO_2-eq, energy use of 9329 MJ, and land use of 32 m^2. Energy use is the main contributor to the direct environmental impact of larvae meal production. The industrial process to acquire larvae meal, however, is still advancing which has potential to increase its energy efficiency. Future development of larvae rearing should, therefore, focus on energy savings to ensure the environmentally sustainability of larvae meal as livestock feed.

Production of larvae meal, however, also has indirect environmental consequences, i.e. environmental impacts related to changes in use of farm inputs or outputs produced. Adding the indirect environmental changes to the direct impact resulted in a GWP of 3132 CO_2-eq, an EU of 36513 MJ, and an LU of 66 m^2. Overall, using 1 ton of larvae meal compared with using 0.5 ton of fishmeal and 0.5 ton of SBM on a DM basis resulted in an increased GWP of 1959 kg CO_2-eq (i.e. excluding LUC) or 1364 kg CO_2-eq (i.e. including LUC), EU of 21342 MJ, and LU decreased with 1713 m^2. Results of the indirect environmental impact, however, are situation specific, e.g. in this study food-waste was used for anaerobic digestion. In case food-waste would have been used for, e.g., composting, the energy use and related emission of greenhouse gases might decrease. Furthermore, the industrial process to acquire housefly larvae meal is still advancing, which also offers potential to reduce the energy use and related emissions. Nevertheless, at this moment using larvae meal results in a trade-off between decreased land use and increased global warming potential and energy use. Eventually, however, land scarcity will increase further, whereas opportunities exist to reduce energy use by, e.g., technical innovations or an increased use of solar or wind energy. Larvae meal production, therefore, has potential to reduce the environmental impact of the livestock sector.

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Chapter 6

Consequential life cycle assessment and feed optimization: alternative protein sources in pig diets

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Abstract

Purpose. Feed production is responsible for the majority of the environmental impact of livestock production, especially for monogastric animals, such as pigs. Several studies demonstrated that replacing soybean meal (SBM) with alternative protein sources, such as locally produced peas or rapeseed meal, potentially reduces the environmental impact of pork production. These studies, however, used an attributional life cycle assessment (ALCA), which solely addresses the direct environmental impact of a product. A replacement of SBM with alternative protein sources, however, can also have indirect environmental consequences, e.g. impacts related to replacing the original function of the alternative protein source. Accounting for indirect environmental consequences in a consequential life cycle assessment (CLCA) might change environmental benefits of using alternative protein sources. This study aims to explore differences in results when performing an ALCA and a CLCA to reduce the environmental impact of pig production. We illustrated this for two case studies: replacing SBM with rapeseed meal (RSM), and replacing SBM with waste-fed larvae meal in diets of finishing-pigs.

Methods. We used an ALCA and CLCA to assess global warming potential (GWP), energy use (EU) and land use (LU) of replacing SBM with RSM and waste-fed larvae meal, for finishing-pigs. The functional unit was one kg of body weight gain.

Results and discussion. Based on an ALCA, replacing SBM with RSM showed that GWP hardly changed (3%), EU hardly changed (1%), but LU was decreased (14%). ALCA results for replacing SBM with waste-fed larvae meal showed that EU hardly changed (1%), but GWP (10%) and LU (56%) were decreased. Based on a CLCA, replacing SBM with RSM showed an increased GWP (15%), EU (12%), and LU (10%). Replacing SBM with waste-fed larvae meal showed an increased GWP (60%) and EU (89%), but LU (70%) was decreased. Furthermore, results showed that assumptions required to perform a CLCA, such as definition of the marginal product, have a large impact on final results.

Conclusion. The CLCA results were contradictory compared with the ALCA results. CLCA results for both case studies showed that using co-products and waste-fed larvae meal not reduces the net environmental impact of pork production. This would have been overlooked when results were only based on ALCA.

Recommendations: To gain insight into the environmental impact of feed, animal nutritionists can use an ALCA. If policy makers or the feed industry, however, want to assess the net environmental impact of a potential mitigation strategy, it is recommended to perform a CLCA. The framework developed in this thesis can be used to perform such an

assessment. Such a CLCA should be based on several scenarios (including different marginal products) to provide different potential pathways to show the range of possible outcomes.

1 Introduction

The global demand for animal source food (ASF) is expected to increase. Simultaneously, livestock production causes severe environmental pressure via emissions to air, water, and soil (Steinfeld et al., 2006). The global livestock sector is responsible for about 15% of the total anthropogenic emissions of greenhouse gases (Gerber et al., 2013). The sector also increasingly competes for scarce resources, such as land, water, and fossil-energy.

Feed production is responsible for the largest part of the environmental impact of livestock production, especially for monogastric production systems (De Vries and De Boer 2010; Gerber et al., 2013). To reduce the environmental impact of feed production a widely applied mitigation is to replace feed ingredients with a high environmental impact for ingredients with a lower environmental impact. Studies, exploring the environmental impact of different feed ingredients, demonstrated that diets containing soybean meal (SBM) often result in a large environmental impact (Cederberg and Flysjo 2004; Eriksson et al., 2005; Van der Werf et al., 2005; Weightman et al., 2011). Currently, SBM is the main protein source in pig diets (Vellinga et al., 2009). Cultivation of SBM has a high environmental impact; due to large transport distances; due to a high economic allocation as SBM nowadays drives the production process (Cederberg and Flysjo 2004; Van der Werf et al., 2005; Vellinga et al., 2009); and due to emissions related to land use change (LUC), such as deforestation in South America (Foley et al., 2007; Prudêncio da Silva et al., 2010).

Several studies have assessed the environmental impact of replacing SBM with: 1) locally produced protein (e.g. peas), 2) co-products from the bio-diesel industry (e.g. rapeseed meal (RSM)), or 3) novel protein sources (e.g. waste-fed larvae meal) (Eriksson et al., 2005; Meul et al., 2012; Sasu-Boakye et al., 2014; Van Zanten et al., 2015a; Van Zanten et al., 2015b). Replacing SBM with locally produced protein sources resulted in a reduction of global warming potential (GWP) up to 13% and of land use (LU) up to 11% (Eriksson et al., 2005; Meul et al., 2012; Sasu-Boakye et al., 2014). Replacing SBM with RSM resulted in a reduction of LU up to 12%, but did not change GWP and energy use (EU) (Van Zanten et al., 2015b). Replacing SBM with waste-fed larvae meal in diets of finishing-pigs is not analysed so far, but preliminary results of Van Zanten et al. (2015a) showed waste-fed larvae meal has potential to reduce the environmental impact of feed production.

Replacing SBM with alternative protein sources, therefore, might be a potential mitigation strategy. Those studies, however, were based on an attributional life cycle assessment (ALCA), implying they addressed the direct environmental impact of a product in a status-quo situation. The direct environmental impacts result from the use of resources and emissions of pollutants directly related to the production of one kg of pig meat, such as feed use, diesel for transport, and electricity for heating, at a specific moment in time. Although commonly used in pig production (McAuliffe et al., 2016), an ALCA does not include consequences of a change in diet composition, outside the production chain of pork.

These indirect environmental consequences are related to changes in use of farm inputs or its outputs. Van Zanten et al. (2015a), for example, explored the environmental impact of using waste-fed larvae meal as livestock feed. These larvae were partly fed on food-waste, which was originally used for production of bio-energy. Accounting for these indirect consequences implied including the environmental impact of production of energy needed to replace the original bio-energy function of food-waste in the analysis. Applying this indirect environmental impact assessment method, called consequential LCA (CLCA), might have different outcomes.

To our knowledge there are only a few studies that assessed the difference in results between an ALCA and a CLCA, when assessing the environmental impact of changing livestock diets (Mogensen et al., 2010; Nguyen et al., 2013), none of them relates to pig production. This study, therefore, aims to explore the differences in results when performing an ALCA and a CLCA to reduce the environmental impact of pig production. We illustrated this comparison with two case studies: replacing SBM with RSM, and replacing SBM with waste-fed larvae meal in diets of finishing-pigs. In this study, GWP, EU, and LU were assessed per kg of body weight gain.

2 Material and method

An ALCA and a CLCA will be used to assess the environmental impact of replacing SBM with RSM or with waste-fed larvae meal. For this purpose, we first describe the diet formulation and growth performance of finishing-pigs (2.1), and subsequently explain the environmental impact assessment methods used (2.2).

2.1 Diet composition and growth performance

All diets were designed to meet the requirements of a Dutch average standard diet for finishing-pigs, and contained 9.50 MJ net energy (NE) and 7.59 g standard ileal digestible (SID) lysine per kg of feed, while pigs were fed ad libitum. Diets had to meet requirements for SID methionine and cystine 62%, SID threonine 65%, and SID tryptophan 20%, relative to SID lysine. Furthermore, because of nutritional reasons and taste, the following dietary restrictions were applied in all scenarios: a diet contained a maximum of 30% maize, 40% wheat, 40% barley, 10% peas, 2% molasses, contained 500 FTU phytase per kg, and 0.4% premix to provide minerals and vitamins.

Given the above mentioned restrictions, the basic scenario (S1) was defined, using SBM as major protein source (see Table 1), based on Van Zanten et al. (2015b). In the second scenario (S2), SBM was replaced with RSM based on their crude protein (CP) content, as described in detail by Van Zanten et al. (2015b). In summary, the amount of CP in 15% SBM and 8% barley was replaced with the CP in 23% RSM. In the last diet (S3), 15% SBM was replaced with 15% waste-fed larvae meal also based on their CP content. The final diet was formulated by using a commercial linear programming tool (i.e. Bestmix®, Adifo, Maldegem, Belgium), with the nutritional value of feed ingredients from CVB database (CVB (Dutch feed tables), 2011). Linear programming was used to optimize the diet by minimizing the cost price of the diet. The same pricelist was used as in S1 and S2. The CVB database, however, does not contain information about the nutrient content and digestibility of waste-fed larvae meal. The digestibility coefficient is needed to assess the actual nutritional intake. Because the actual nutritional intake is based on the nutrient content multiplied with the digestibility coefficient. The nutrient content of waste-fed larvae meal (Table 2) was adapted from Van Zanten et al. (2015a), but values were consistent with a literature review of Makkar et al. (2014). Information about the digestibility coefficient of waste-fed larvae meal for pigs is unknown. Information about the digestibility coefficient of waste-fed larvae meal for poultry is, however, available. As the digestibility coefficient for poultry and pigs is quite similar for other protein-rich ingredients, such as SBM and fishmeal, calculation on the digestibility coefficient of waste-fed larvae meal were based on the digestibility coefficient for poultry (Appendix Table A.1 and Table A.2). By using the following equation (CVB, 2011), the net energy (NE) value of waste-fed larvae meal was calculated resulting in 13.01 MJ per kg waste-fed larvae meal:

$$
\begin{aligned}
&NE\ (kJ/kgDS) = \\
&(10.8\ x\ 425\ digestible\ crude\ protein) + (36.1\ x\ 228\ digestable\ crude\ fat) + \\
&(13.7\ x\ 0\ starch) + (12.4\ x\ 0\ sugar) + (9.6\ x\ 20\ remaining\ carbohydrates).
\end{aligned}
$$

As the nutrient content of the diet in each scenario was identical (9.50 MJ NE/kg feed and 7.59 g lysine/kg feed), and no adverse effect of pig performance were found by including RSM (McDonnell et al., 2010) or waste-fed larvae meal (Makkar et al., 2014) in finishing-pig diets, a similar growth performance was assumed between the three scenarios. Growth performance was based on Van Zanten et al. (2015b), who calculated the growth performance of finishing-pigs for S1 and S2. Scenarios started with 100 days, weight at start 45 kg, final age 180 days, and total feed use 183 kg. The final body weight of the growing-pigs was 116.4 kg (Van Zanten et al., 2015b).

Table 1. Diet composition of scenario 1 (S1) containing SBM, scenario 2 (S2) containing RSM, and scenario 3 (S3) containing larvae meal.

Ingredients	**S1**	**S2**	**S3**
Rapeseed meal, CP <380	-	23.00	-
Soybean meal, CP<480	15.00	-	-
Waste-fed larvae meal	-	-	15.00
Peas	9.36	10.00	-
Maize	30.00	30.00	30.00
Wheat	29.74	30.24	24.29
Wheat middlings	0.90	-	26.57
Barley	10.10	-	-
Sugarcane molasses	2.00	2.00	2.00
Vit. and min. premix	0.40	0.40	0.40
Phytase premix	0.65	0.65	0.65
Animal fat	-	2.09	-
Limestone	1.24	0.96	1.10
Salt	0.37	0.29	0.26
Monocalcium phosphate	0.11	0.01	-
Sodium bicarbonaat	-	0.09	0.15
L-Lysine HCL	0.10	0.22	0.03
L-Tryptophan	-	0.01	-
L-Threonine	-	0.02	-
DL-Methionine	0.03	0.01	-
Nutrient content g/kg			
Nett energy, MJ	9.5	9.5	9.5
Crude protein	162	160	166
Lysine (SID)	7.59	7.59	7.59
Crude fibre	30	47	45
Crude fat	27	50	60
Phosphorus (P)	3.75	4.65	5.31
Digestible P	2.27	2.27	2.27

Table 2. Nutrient content (g/kg) of soybean meal (SBM) and rapeseed meal (RSM) based on CVB (2010), and waste-fed larvae meal (a) based on data of a laboratory plant (Van Zanten et al., 2015a) and waste-fed larvae meal (b) based on the average value found in Makkar et al. (2014).

	SBM	RSM	Larvae meal (a)	Larvae meal (b)
Dry matter	87.3	87.3	88.0	-
Crude protein	46.4	33.5	47.9	50.4 ± 5.3
Crude fat	1.9	2.6	24.2	-
Crude fibre	3.7	12.0	6.4	5.7 ± 2.4
Ash	6.5	6.7	6.2	10.1 ± 3.3
Phosphorus	0.6	1.1	9.5	16.0 ±5.5
Calcium	0.3	0.7	8.5	4.7 ± 1.7
Lysine (g/16gN)	6.2	5.5	6.8	6.1 ± 0.9
Methionine (g/16gN)	1.4	2.0	2.4	2.2 ± 0.8

2.2 Life cycle assessment

To assess the environmental impact of each scenario, a life cycle assessment (LCA) was used. LCA is an internationally accepted and standardized method (ISO14040, 1997; ISO14041, 1998; ISO14042, 2000; ISO14043, 2000) to evaluate the environmental impact of a product during its entire life cycle (Guinée et al., 2002; Bauman and Tillman, 2004). During the life cycle of a product two types of environmental impacts are considered: emissions of pollutants and use of resources, such as land or fossil-fuels (Guinée et al., 2002). We assessed GHG emissions, EU, and LU. These impacts were chosen because the livestock sector contributes significantly to both LU and climate change worldwide (Steinfeld et al., 2006a). Furthermore, EU was used as it influences GWP considerably. LU was expressed in m^2 per year, whereas EU was expressed in MJ. The following GHGs were included: carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O). These GHGs were summed up based on their equivalence factors in terms of CO_2 (100 years' time horizon): i.e. carbon dioxide (CO_2), biogenic methane (CH_4, bio): 28 kg CO_2-eq/kg, fossil methane (CH_4, fossil): 30 kg CO_2-eq/kg; and nitrous oxide (N_2O): 265 kg CO_2-eq/kg (Myhre et al., 2013). In this study only the environmental impact related to feed production is assessed because no changes are expected on related emissions of piglet production (rearing), enteric fermentation from pigs, and from pig housing. Changes from manure management can be expected but no data is available on related emissions of manure management when insects are used as feed.

As stated in the introduction two types of LCA exist: ALCA and CLCA. Both methods are explained below.

Attributional LCA

An ALCA describes the environmentally relevant physical flows to and from all processes, in the life cycle of a product, at one specific moment in time. During the life cycle of a product, like pork, multifunctional processes occur. A multifunctional process (also referred as product-packages) is an activity that fulfils more than one function (Ekvall and Weidema, 2004), yielding two or more products: the determining product, which determines the production volume of that process (e.g. rapeseed oil), and a co-product (e.g. rapeseed meal; Weidema et al., 2009). In case of a multifunctional process, most ALCA studies of livestock products partition the environmental impact of the process to the various products based on their relative economic values, a method called economic allocation (De Vries and De Boer, 2010). In our ALCA, we used economic allocation to divide the environmental impact between the determining product and the co-product.

To assess the environmental impact of the three scenarios, the environmental impact of each ingredient must be known. GWP, EU, and LU of most feed ingredient were based on Vellinga et al. (2013). Production of feed ingredients included impacts from cultivation (e.g. impacts related to the production and use of fertilizers, pesticides, machinery, and energy), impacts from drying/processing, and impacts from transport to the farm. GWP, EU, and LU related to waste-fed larvae meal were based on Van Zanten et al. (2015a). LU and EU values of feed additives (salt, chalk, vitamins and minerals, phytase, monocalcium phosphate, and amino acids) were based on Garcia-Launay et al. (2014) (GWP was based on Vellinga et al. (2013)). Appendix Table A.3 provides an overview of GWP, LU, and EU per kg of feed ingredient. To assess the average impact of one kg feed, the environmental impact per kg feed ingredient was multiplied by it relative use in the diet. Next, for each scenario, the average environmental impact per kg feed was multiplied with the total feed intake during the finishing period and divided by the growth performance during the finishing period (116.4 kg – 45 kg = 71.4 kg). The functional unit was one kg weight gain.

Consequential LCA

A CLCA describes how environmental flows change in response to a change in the system (Ekvall and Weidema, 2004). Only those processes (within and outside the system) that respond to the change, are considered. Considering changes is especially important when a mitigation strategy includes the use of co-products or food-waste. This is because the production volume is restricted for co-products and food-waste. For co-products, for example, a change in demand of the determining product (e.g. sugar) directly affects the production volume of the co-product (e.g. beet pulp) (Weidema et al., 2009), whereas a change in demand of co-product does not. Due to this, co-products are limited available.

Increasing the use of co-products in animal feed, therefore, results in a reduction of co-product use in another sector necessitating substitution (Appendix A.1).

Within CLCA, system expansion is generally used to handle multifunctional processes. System expansion implies that you include changes in the environmental impact of the alternative production process for which the co-product could be used, into your analysis (Ekvall and Finnveden, 2001). Van Zanten et al., (2014) developed a theoretical framework to assess the environmental consequences of using co-products in livestock feed. This framework provides assistance in how to assess the environmental impact of changing the application of a co-product. In this study, the theoretical framework of Van Zanten et al. (2014) was used to assess the environmental consequences of replacing SBM with RSM or with waste-fed larvae meal. Based on this framework the net environmental impact was calculated. The net environmental impact depends on the environmental benefits minus the environmental costs. The environmental benefits are determined by the decrease in environmental impact related to the product that was replaced with co-products or food-waste). The environmental costs are determined by the increased environmental impact related to the marginal product (the product that replaces the 'old' application of the co-product or food-waste).

To assess the consequences of replacing SBM with RSM or waste-fed larvae meal in the diet of finishing-pigs, the following steps were needed.

First, the difference in feed ingredients between the diet in S2 and S3, and the reference scenario (S1) was determined, by subtracting all feed ingredients used in S1, from those in S2 and in S3 (see Table 3). Table 3 shows which feed ingredients changed compared with the basic scenario containing SBM, for example, replacing SMB with RSM, i.e. resulted in an increase in RSM of 23%, a decrease of 15% SBM etcetera. In case a feed ingredient is used in the same amount, such as maize, it is not considered as it does not result in an environmental change.

Second, the environmental impact of a change in each feed ingredient was determined (Table 3). The computation of this environmental impact differed depending on the feed ingredient being: a determining product without a co-product (so no product-package); or a determining product with a co-product; or a co-product. In case a feed ingredient is a determining product and not part of a product-package (e.g. peas), the environmental impact of that determining product was based on ALCA data of Vellinga et al. (2013). This was because when a product is not part of a product-package, no additional environmental consequences occur. In case a feed ingredient is part of a product-package, the environmental consequences related to the ingredient were calculated based on the principles of the theoretical framework described by Van Zanten et al. (2014).

In case SBM was replaced with RSM (i.e. comparing S1 with S2) the products that were part of a product-package were: SBM (i.e. a determining product with oil as co-product) and the co-products wheat middlings, RSM, and animal fat. In case waste-fed larvae meal replaced SMB (i.e. comparing S1 with S3) the products that were part of a product-package were: SBM and the co-products wheat middlings, and waste-fed larvae meal. The indirect environmental consequences related to waste-fed larvae meal was based on Van Zanten et al. (2015a). For the other four products - RSM, animal fat, wheat middlings, and SBM - the environmental impact was calculated. Figure 1 illustrates the calculation of the environmental impact of an example case, namely RSM. RSM is a co-product from the bio-diesel industry, and does not drive the production process. An increased use of RSM in diets of finishing-pigs, therefore, results in a reduction of the original applications of RSM. We assumed that RSM was originally used in diets of dairy cows. Increasing the use of RSM in pig diets, therefore, resulted in a decreased use of RSM in diets of dairy cow. RSM in diets of dairy cows, therefore, was replaced (or also often called displaced) by the marginal product, which we assumed to be SBM (Weidema, 2003). Replacing RSM with SBM in diets of dairy cows was based on net energy for lactation, as this was the limiting nutritional factor of SBM. An increased production of SBM also results in an increased production of soy-oil, the depended co-product. The increased production of soy-oil was assumed to replace the marginal oil, being palm-oil (Dalgaard et al., 2008). A reduction in production of palm-oil, however, also implies a reduction in production of palm kernel meal. A reduction of palm kernel meal resulted in an increased use of the marginal meal SBM. Replacing palm kernel meal with SBM was based on their energy and protein content, as suggested by Dalgaard et al. (2008). The reduction of 19 g of palm kernel meal, therefore, was replaced with 3 g SBM and 15 g barley. Barley is assumed to be the marginal feed grain (Weidema, 2003). Thus, the amount of CP and energy in palm kernel meal is equal to the total amount of CP and energy in SBM and barley.

Figure 2 illustrates how the environmental consequences of animal fat, wheat middlings, and SBM is calculated. The same principle as for RSM, based on the theoretical framework of Van Zanten et al. (2014), was applied. In the Appendix (A.2) the calculations related to animal fat, wheat middlings, and SBM are explained in more detail.

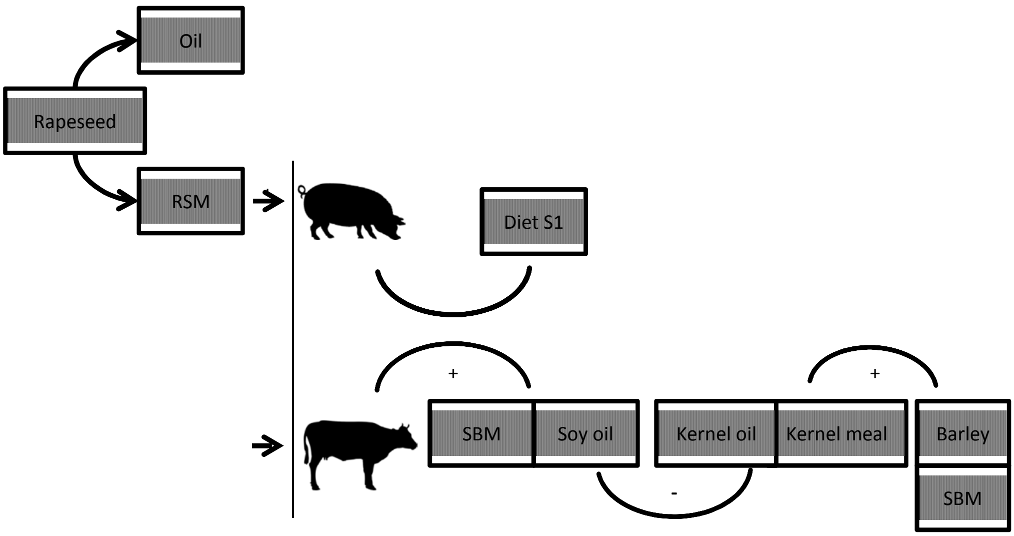

Figure 1. Principle to assess the environmental consequences of rapeseed meal (RSM) based on Van Zanten et al. (2014).

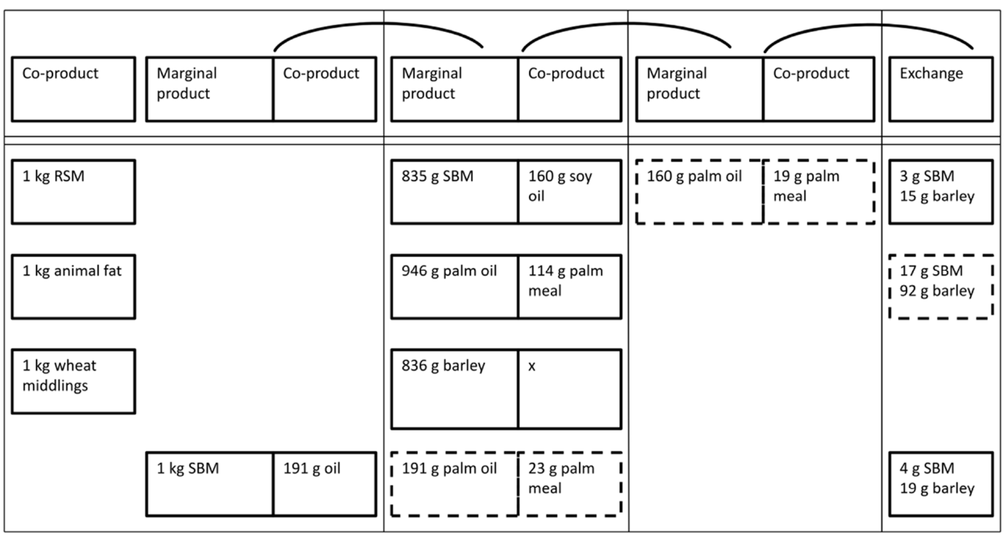

Figure 2. Description of the environmental consequences of increasing rapeseed meal (RSM), animal fat, and wheat middlings and decreasing the use of SBM in diets of finishing-pigs. The full-lines represent an increased production of a product while the dotted-lines represent a decreased production of a product.

Table 3. Global Warming Potential (GWP) expressed in g CO_2-eq per kg of final diet, energy use (EU) expressed in MJ per kg of final diet, and land use (LU) expressed in m^2 per kg of final diet of replacing soybean meal (S1) with rapeseed meal (S2) and replacing soybean meal (S1) with waste-fed larvae meal (S3) in pig diets.

	S1 - S2	**GWP**	**EU**	**LU**	***S1 – S3***	**GWP**	**EU**	**LU**
	g/kg	CO_2-eq	MJ	m^2	*g/kg*	CO_2-eq	MJ	m^2
Rapeseed meal	*230.0*	289	2.4	4.83	*0.0*	-	-	-
Soybean meal	*150.0*	267	2.5	0.57	*-150.0*	267	2.5	0.57
Larvae meal	*0.0*	-	-	-	*150.0*	3068	36.5	0.07
Peas	*6.4*	741	6.6	5.71	*-93.6*	741	6.6	5.71
Maize	*0.0*	-	-	-	*0.0*	-	-	-
Wheat	*5.0*	378	3.0	1.14	*-54.5*	378	3.0	1.14
Wheat middlings	*-9.0*	387	3.4	1.07	*256.7*	387	3.4	1.07
Barley	*101.0*	379	2.9	1.28	*-101.0*	379	2.9	1.28
Sugarcane molasse	*0.0*	-	-	-	*0.0*	-	-	-
Premix	*0.0*	-	-	-	*-2.0*	4999	0.8	0.00
Phytase premix	*0.0*	-	-	-	*-2.0*	4999	26.0	0.15
Fat from animals	*20.9*	4828	21.53	1.76	*0.0*	-	-	-
Chalk	*-2.8*	19	0.00	0.00	*-1.4*	19	0.0	0.00
Salt	*-0.8*	180	3.50	0.00	*-1.1*	180	3.5	0.00
Monocalcium-Phosphate	*-1.0*	4999	18.4	0.32	*-1.1*	4999	18.4	0.32
Bicarbonaat	*0.9*	180	3.9	0.00	*1.5*	180	3.9	0.00
L-Lysine HCL	*1.2*	6030	119.9	2.27	*-0.7*	6030	119.9	2.27
L-Threonine	*0.2*	16978	119.9	2.27	*0.0*	-	-	-
DL-Methionine	*-0.2*	5490	89.3	0.01	*-0.3*	5490	89.3	0.01

6

3 Results

Using an ALCA approach, S1 resulted in 1.62 kg CO_2-eq, 14.01 MJ, and 4.81 m^2.yr per kg weight gain, S2 in 1.67 kg CO_2-eq, 14.11 MJ, and 4.12 m^2.yr per kg weight gain, and S3 in 1.45 kg CO_2-eq, 14.17 MJ, and 2.14 m^2.yr per kg weight gain.

Replacing SBM (S1) with RSM (S2) based on an ALCA, therefore, hardly changed GWP and EU but it decreased LU per kg weight gain, implying that this strategy has no potential to reduce GWP and EU but has potential to reduce LU of pork production. Using a CLCA approach, this strategy resulted in an increase of 0.25 kg CO_2-eq, 1.61 MJ, and 0.48 m^2.yr per kg weight gain, yielding even less unambiguous results. Relative differences between the ALCA and CLCA approach are presented in Figure 3.

Replacing SBM (S1) with waste-fed larvae meal (S2) based on an ALCA approach resulted in a decreased GWP and LU and hardly changed EU, implying this strategy has potential to reduce GWP and LU but has no potential to reduce EU of pork production. Using a CLCA approach, this strategy resulted in an increase of 0.97 kg CO_2-eq, 12.51 MJ, and a reduction of 3.38 m^2.yr per kg weight gain, yielding less unambiguous results. Relative differences between the ALCA and CLCA approach are presented in Figure 4.

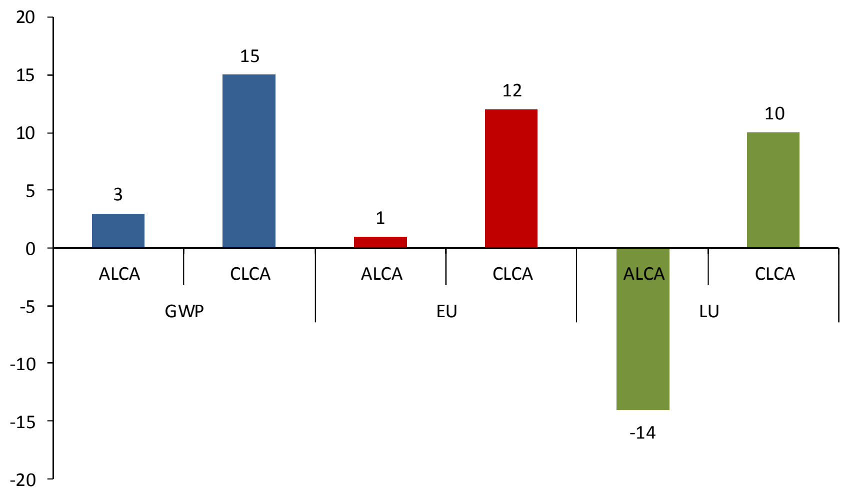

Figure 3. The environmental impact of replacing SBM with RSM in pig diets based the attributional LCA approach and the consequential LCA approach in %.

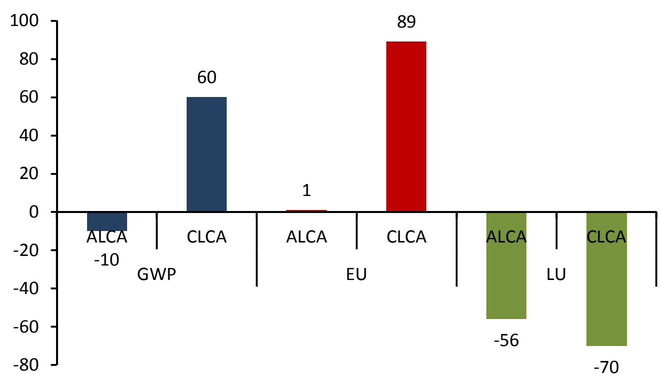

Figure 4. The environmental impact of replacing SBM with waste-fed larvae meal in pig diets based on the attributional LCA approach and the consequential LCA approach in %.

4 Discussion

4.1 Differences between ACLA results and CLCA results

Based on an ALCA, replacing SBM with RSM reduced LU, but hardly changed GWP and EU. Based on a CLCA, however, replacing SBM with RSM resulted in an increased GWP, EU, and LU. Differences in results between ALCA and CLCA were caused because the net environmental impact was increased. In S1 15% SBM and 8% barley was replaced with 23% RSM, 2% animal fat. As RSM and animal fat are both co-products, using them resulted in indirect environmental consequences. The increased impact (environmental costs) related to the consequences of using co-products (using RSM resulted in an increased use of SBM in diets of dairy cows, whereas using animal fat resulted in an increased use of palm oil in broiler diets) was higher compared to the reduction in impact (environmental benefits) due to decreasing SBM and barley in pig diets.

Based on an ALCA, replacing SBM with waste-fed larvae meal reduced GWP and LU, but hardly affected EU. Based on a CLCA replacing SBM with waste-fed larvae meal resulted in an increased GWP and EU but LU was still decreased. The difference in results between ALCA and CLCA were mainly caused by the high environmental impact of the waste-fed larvae meal. The environmental impact of waste-fed larvae meal was based on Van Zanten et al. (2015a). Larvae were fed partly on food-waste. Food-waste was originally used for anaerobic digestion in the Netherlands. Using food-waste for waste-fed larvae meal production decreased its availability for anaerobic digestion, as the amount of food-waste was limited by the amount of food spilled by humans. The decreased production of electricity, heat and digestate, therefore, was substituted by fossil-fuels and synthetic fertilizer, resulting in an increased environmental impact. Although waste-fed larvae meal can be a feed ingredient with a high nutritional value, changing the application of food-waste, from bio-fuel production to waste-fed larvae meal production, does not reduce the overall environmental impact.

Results of both case studies, therefore, showed that using co-products and food-waste not necessarily results in a reduction of the environmental impact.

4.2 Impact of assumptions

Replacing SBM with RSM or waste-fed larvae meal resulted in an increased net environmental impact. We made assumptions that were plausible for the current situations. World food and feed markets systems are, however, highly complex and dynamic, which might change the assumptions we made. To what extend did the assumptions made affect the final results? Our main assumptions related to 1) different livestock species 2) nutritional value of the feed ingredients, and 3) marginal product.

Let's discuss the choice of livestock species first. We assumed, based on Dutch practice, that RSM and wheat middlings were used in diets of dairy cows, whereas animal fat was used in diets of broilers. Based on the nutritional value of SBM compared to RSM, however, dairy cows need more SBM compared to broilers to replace one kg of RSM. Dairy cows also need more barley to replace one kg of wheat middlings compared to broilers, while they need less palm oil to replace one kg animal fat. In case livestock species are, therefore, shifted (RSM and wheat middlings were used in broiler diets and animal fat in diets of dairy cows) results will change. GWP, EU, and LU decreased in both scenarios (see Appendix A.3 for calculations).

Second, assumptions related to nutritional value of the feed ingredients. Replacing RSM with SBM, and wheat middlings with barley was based on their energy value, as energy was the limiting factor. It should, however, be noted that besides energy those ingredients also provide protein. In case the replacement is based on protein GWP, EU, and LU decreased for both scenarios (see Appendix A.3 for calculations). Replacement based on the limiting factor (energy), therefore, will result in an overestimation, while replacement based on protein will result in an underestimation. Replacement based on either energy or protein is, therefore, restrictive. This is exactly the reason why we assessed the replacement of SBM with RSM or waste-fed larvae meal in pig feed on diet level instead of feed ingredient level.

Last, assumptions related to the marginal product. RSM, wheat middlings, animal fat, and bio-diesel were replaced with a marginal product. The marginal product is the product that responds to a change in demand. RSM was assumed to be replaced with SBM, wheat middlings with barley, animal fat with palm oil, and bio-diesel with fossil-fuel. The marginal product is, however, subject to numerous socio- and economic aspects. The marginal product, for example, can change over time and can differ across space. Worldwide, food-waste, for example, is not used for anaerobic digestion but ends up in landfills, or is used for composting or as animal feed. Changing the assumption related to the marginal product affects the results. In case, for example, the marginal energy source was wind- or solar-energy, instead of fossil-fuel, using waste-fed larvae meal might decreased the environmental impact (Van Zanten et al. 2015a). The results of a CLCA, therefore, largely depend on the assumptions made related to the marginal product (Schmidt and Weidema, 2008; Reinhard and Zah, 2011; Dalgaard et al., 2014). As a change in a system can result in different possible chain-of-event pathways, CLCA results are rather uncertain (Thomassen et al., 2008; Nguyen et al., 2013; Plevin et al., 2014).

4.3 ALCA versus CLCA

Assessing the status quo of a pig system by performing an ALCA can create understanding about the environmental impact of the current situation, and can yield hotspots (i.e. processes with a major impact), and potential improvement options. By performing an ALCA, we identified that replacing SBM with RSM or waste-fed larvae meal can (partly) reduce the environmental impact. Results of this study, however, also showed that an ALCA does not grasp the full complexity of the consequences of implementing an innovation. Based on the results of an ALCA study, one can easily conclude that feeding more co-products or waste products to livestock results in an improved environmental impact, while this is not necessarily the case.

To assess the environmental impact of implementing an innovation, a CLCA is suitable, especially in combination with scenario analysis to underpin uncertainty (Zamagni et al., 2012; Plevin et al., 2014; Meier et al, 2015). By performing a CLCA, information will be provided on the environmental change in comparison with the current situation. CLCA studies can be highly relevant especially in case we assess the environmental impact of a novel feed ingredient. Such studies provide information about interactions outside the production chain resulting in environmental consequences when the innovation will be implemented, providing support to policy makers during decision making (Zamagni et al., 2012; Plevin et al., 2014; Meier et al., 2015).

The difficulty of performing a CLCA related feed optimization, however, is that it requires insight into world food and feed markets. Especially in pig feed, where feed optimization is based on least cost optimization and a wide range of ingredients are available, diet formulations can change from day to day resulting each in different environmental impacts. Currently, we see developments as precision farming in which each finishing-pig is fed based on it individual needs. It is hard to fully grasp such a complex and dynamic system with an LCA (Plevin et al., 2014). Some researcher as in the report of Dalgaart et al. (2007), therefore, 'simplify' the assumptions and state that at the end an increased demand for pork always results in an increased demand for the marginal protein source, SBM, and the marginal energy source, barley, although different feed ingredients are used. We can, however, wonder if this is correct? The starting point of a CLCA is the point where the stone hits the water, resulting in waves, the so called cause-and-affect chain (Ekvall and Weidema, 2004). The first waves, or first consequences have most impact. Ekvall and Weidema (2004), therefore, advise to include only the environmental relevant waves and not to estimate consequences far down the cause-and-effect chain. In this study we experienced that performing a CLCA of a complete diet resulted in many consequences on different levels (some far down the cause-and-effect chain). Using RSM in pig diets, for example, increased SBM in diets of dairy cows, resulting in a decrease of palm oil, resulting in a decrease of palm kernel meal, resulting eventually in an increase of SBM. Such consequences will also affect feed prices, resulting eventually in different feed optimization with again different environmental impacts. This complexity makes it difficult to get reliable results when a CLCA for a single diet is assessed, as uncertainties related to the cause-and-effect chain are high.

Results of this study showed, furthermore, that diet formulations are complex, and simplifying the assumptions does not provide e.g. feed companies or policy makers insight on how to reduce the environmental impact of their diets. For example, the study of Van Zanten et al. (2015a) assumed based on CP content, that waste-fed larvae meal will replace SBM or fishmeal. We found, however, based on feed optimization that waste-fed larvae meal replaces protein sources (SBM) and energy sources (barley) and consequently more co-products e.g. wheat middlings were used.

Related to feed optimization, we recommend farmers and animal nutritionist to use an ALCA to get insight in the environmental impact of their feed. In case, however, a policy maker or the feed industry wants to apply a mitigation strategy, it is recommended to perform a CLCA. Such a CLCA should be based on several scenarios (e.g. including different marginal products) to provide insight into different pathways.

5 Conclusion

Based on an ALCA, replacing SBM with RSM reduced LU, but did not affect GWP and EU. Whereas replacing SBM with waste-fed larvae meal decreased GWP and LU, but did not affect EU. Based on a CLCA, replacing SBM with RSM increased impacts on the environment. Replacing SBM with waste-fed larvae meal resulted in an increased GWP and EU but still reduced LU. The CLCA results were, therefore, contradictory with the standard ALCA results. Environmental benefits from an ALCA appeared more promising than from a CLCA. CLCA results for both case studies showed that using co-products and food-waste not necessarily reduces the environmental impact of pork production. For both cases, replacing SBM with RSM or waste-fed larvae meal resulted in an increased net environmental impact. This would have been overlooked when results were only based on ALCA.

Furthermore, results of this study showed that assumptions required to perform a CLCA, such as definition of the marginal product, have a large impact on final results. Results of a CLCA, therefore, seem to be relatively more uncertain compared to results of the ALCA, but more exact. Related to feed optimization, we recommend animal nutritionists to use an ALCA to get insight in the environmental impact of their feed. If policy makers or the feed industry want to assess the environmental benefits of a mitigation strategy, however, it is recommended to perform a CLCA. Such a CLCA should be based on several scenarios (e.g. including different marginal products) to provide insight into different potential pathways and decrease uncertainty.

6 Acknowledgements

The authors thank Wageningen UR Livestock Research for funding this study.

A Appendix

Table A.1. Apparent total tract digestibility coefficients of house fly larvae determined in studies with different types of poultry.

Source	**Hwangbo et al. (2009)**	**Zuidhof et al. (2003)**	**Pretorius (2011)**	**Mean value**
Animal	Broiler	Turkey	Broiler	
Components				
Crude protein	98.5	98.8	69.0	88.8
Crude fat	-	-	94.0	94.0
Crude fiber	-	-	62.0	62.0
Amino acids				
Arginine	95.5	91.7	-	93.6
Alanine	95.7	94.4	-	95.1
Aspartic acid	93.2	93.2	-	93.2
Cystine	92.7	78.1	-	85.4
Glutamic acid	95.1	93.9	-	94.5
Glycine	95.5	88.0	-	91.8
Histidine	93.7	94.3	87.0	91.7
Isoleucine	92.2	93.9	-	93.1
Leucine	94.7	93.5	-	94.1
Lysine	97.6	96.9	-	97.3
Methionine	95.6	97.7	-	96.7
Phenylalanine	96.8	96.5	-	96.7
Proline	93.4	89.7	-	91.6
Serine	95.6	91.0	-	93.3
Threonine	93.3	91.3	93.0	92.5
Tryptophan	93.9	93.1	95.0	94.0
Tyrosine	96.1	98.0	-	97.1
Valine	94.5	93.8	91.0	93.1

Table A.2. Comparison of the digestibility coefficient (in %) for crude protein, crude fat, and amino acids (AID) between pigs and broilers for soybean meal (SBM) and fishmeal (CVB, 2011).

Ingredient	**SBM**	**SBM**	**fishmeal**	**Fishmeal**
Animal	Pigs	Broiler	Pigs	Broiler
Total tract				
Crude protein	93	85	87	86
Crude fat	65	71	87	87
Ileaal for pigs and total tract for broilers				
Arginine	93	89	91	92
Alanine	85	83	89	91
Aspartic acid	87	89	77	83
Cystine	82	82	70	89
Glutamic acid	90	91	89	89
Glycine	83	81	85	84
Histidine	89	89	85	84
Isoleucine	87	88	89	89
Leucine	87	88	89	91
Lysine	89	88	89	90
Methionine	90	88	88	91
Phenylalanine	88	89	86	89
Proline	89	89	94	84
Serine	87	88	87	84
Threonine	84	85	86	85
Tryptophan	86	89	86	85
Tyrosine	88	89	86	88
Valine	86	87	88	91

Table A.3. Global Warming Potential (GWP) expressed in g CO_2-eq per kg product, energy use (EU) expressed in MJ per kg product, and land use (LU) expressed in m^2.yr per kg product based on the attributional LCA approach.

Ingredients	GWP	EU	LU
Rapeseed, extruded	456	3.4	1.25
Soybean meal	694	5.9	3.11
Larvae meal	785	9.3	0.00
Peas	741	6.6	5.71
Maize	580	5.2	1.29
Wheat	378	3.0	1.14
Wheat middlings	243	2.2	0.58
Barley	379	2.9	1.30
Sugarcane molasses	319	3.7	0.22
Phytase premix	4999	26.0	0.15
Mervit starter 2220	4999	0.8	0.00
Animal fat	823	12.4	0.00
Limestone	19	0.0	0.00
Salt	180	3.5	0.02
Monocalcium phosphate	4999	18.4	0.32
Sodium bicarbonaat	180	3.9	0.00
L-Lysine HCL	6030	119.9	2.27
L-Threonine	16978	119.9	2.27
DL-Methionine	5490	89.3	0.01

A1 Attributional LCA and consequential LCA related to co-products

Feeding livestock mainly co-products from arable production or the food processing industry offers potential to reduce the environmental impact of livestock products, such as pork, chicken, and eggs. The amount of co-products available, however, is limited and dependent on the production volume of the determining product. For example, the amount of wheat middlings depends on the production volume of wheat flour. This means that when company A decides to increase its use of co-products in livestock diets, fewer co-products are available for company B, which has to adapt his production plan. Based on an ALCA, which does not take the consequences for company B into account, increasing the amount of co-products is a promising strategy to reduce the environmental impact of company A. However, taking into account the consequences for company B, might give a different result: the environmental benefit of increasing the use of co-products in company A will depend on the current application of the co-product in company B. By performing a CLCA, information will be provided on the environmental consequences in comparison with the current situation. So, if the current application of a co-product is bio-energy, and the new application will be livestock feed, the consequences related to the decrease in bio-energy production will be taken into account.

Note: explanation is based on the book chapter ‘Future of animal nutrition: the role of life cycle assessment’ by C.E. van Middelaar, H.H.E. van Zanten, I.J.M. de Boer in ‘Sustainable nutrition and feeding of pigs and poultry’ which will be published soon.

A2 Calculation the environmental impact of wheat middlings, animal fat, and SBM based on the theoretical framework of Van Zanten et al. (2014)

Wheat middlings. An increased use of the co-product wheat middlings in diets of finishing-pigs resulted in a reduction of the original application. We assumed that wheat middlings were originally used in diets of dairy cows and that wheat middlings were replaced with barley (the marginal product). The replacement of wheat middlings with barley in diets of dairy cows was based on energy content of barley. An increased production of barley resulted also in an increased production of straw. Straw can be used as bedding material but eventually should be returned to the field to prevent depletion of soil organic matter. We, therefore, did not take straw into account.

Animal fat. An increased use of the co-product animal fat in diets of finishing-pigs resulted in a reduction of the original applications. We assumed that animal fat was originally used in broiler diets and that animal fat was replaced with palm oil (the marginal product). The replacement of animal fat with palm oil in broiler diets was based on energy content. An increased production of palm oil resulted also in an increased production of palm kernel meal, the depended co-product. Palm kernel meal displaces SBM, the marginal product. The displacement of the marginal product is again based on the energy and protein content and follows the same principles as described in the paper.

SBM. A decreased use of the determining product SBM in diets of finishing-pigs resulted in a reduction of soybean production. A reduced production of SBM resulted also in a reduced production of soybean-oil, the depended co-product. The decreased production of soy-oil increased palm-oil production, the marginal product (Dalgaard et al., 2008).The production of palm-oil yields, however, palm kernel meal as well. Palm kernel meal displaces SBM, the marginal product. The displacement of the marginal meal is again based on the energy and protein content and follows the same principles as described in the paper.

A3 Changing assumption related to the CLCA approach

Assumptions related to different livestock species.

When using RSM and wheat middlings in diets of pigs one accounts for the decreased use of RSM and wheat middlings in diets of dairy cows, resulting in an increased use of SBM and barley. To replace 230 g RSM, for example, dairy cows need 192 g SBM while the 230 g RSM in pig diets only reduced 150 g SBM (difference between S1 and S2). To replace 1 kg RSM, 0.84 kg SBM was needed based on energy content (net energy for lactation of RSM 848 VEM per kg and SBM 1015 VEM per kg) and to replace 1 kg wheat middlings, 0.84 kg barley was needed (net energy for lactation RSM 815 VEM per kg and SBM 975 VEM per kg). In case RSM and wheat middlings were not replaced in diets of dairy cows but in broiler diets, less SBM and barley was needed. To replace RSM 0.76 kg, SBM was needed (ME for poultry in RSM 6.99 per kg and in SBM 9.19 MJ per kg) and to replace wheat middlings, 0.67 kg barley was needed (ME for poultry in wheat middlings 7.72 Oepl per kg and Oepl barley 11.67 per kg). In case animal fat was not replaced in broiler chicken diets but in diets of dairy cows, less palm oil was needed, 0.93 kg palm oil (net energy for lactation of animal fat 3264 VEM per kg and VEM palm-oil 3514 per kg) instead of 0.95 kg to replace 1 kg animal fat (ME for poultry in animal fat 35.47 Oepl per kg and palm oil 37.48 Oepl per kg If livestock species are shifted in S1-S2 GWP decreased from 15% to 14%, EU from 12% to 11%, and LU from 10% to 5%. If livestock species are shifted in S1-S3, GWP decreased from 60% to 54%, EU from 89% to 87%, and LU from -70% to -76%. So, GWP, EU, and LU decreased in both scenarios.

Assumptions related to choice of nutrient of the feed ingredients.

The difference in results can be explained by differences in the nutrient value of an ingredient per livestock species. Replacing RSM by SBM in diets of dairy cows was based on the energy content of SBM, as energy was the limiting factor. When 1 kg RSM was replaced based on protein content, however, 0.57 kg SBM was needed (true protein digested in the small intestine RSM 126 g DVE per kg and SBM 221 g DVE per kg). The same occurs when wheat middlings were used. Based on energy, 0.84 kg barley was needed and 0.57 kg barley was needed based on protein to replace 1 kg wheat middlings. If calculations were based on protein content instead of energy in S1-S2, GWP decreased from 15% to 13%, EU decreased from 12% to 9%, and LU from 10% to -9%. If calculations were based on protein instead of energy in S1-S3, GWP decreased from 60% to 54%, EU from 89% to 86%, and LU from -70% to -74%.

Assumptions related to the marginal product.

RSM, wheat middlings, animal fat, and bio-diesel were replaced with the marginal product. The marginal product is the product that response on a change in demand. RSM was replaced with the marginal protein source SBM, wheat middlings by the marginal energy source barley, animal fat by the marginal oil source palm oil, and bio-diesel by the marginal energy source fossil-fuel. Changing the assumption related to the marginal product affects the results. To give another example, if the marginal oil is sunflower oil instead of palm-oil, GWP decrease from 15% to 12%, EU would increase from 12% to 14%, and LU from 10% to 23% for S1-S2. In S1-S3, GWP would decrease from 60% to 50%, EU increased from 89% to 96%, and LU increased from -70% to -33%.

Chapter 7

Global food supply: land use efficiency of livestock systems

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Abstract

Purpose. Livestock already use most global agricultural land, whereas the demand for animal-source food (ASF) is expected to increase. To address the contribution of livestock to global food supply, we need a measure for land use efficiency of livestock systems.

Methods. Existing measures capture different aspects of the debate about land use efficiency of livestock systems, such as plant productivity and the efficiency of converting feed, especially human-inedible feed, into animal products. So far, the suitability of land for cultivation of food-crops has not been accounted for. Our land use ratio (LUR) includes all above-mentioned aspects and yields a realistic insight into land use efficiency of livestock systems. LUR is defined as the maximum amount of human-digestible protein (HDP) derived from food-crops on all land used to cultivate feed required to produce 1 kg ASF over the amount of HDP in that 1 kg ASF. We illustrated our concept for three case systems.

Results and discussion. The LUR for the case of laying hens equalled 2.08, implying that land required to produce 1 kg HDP from laying hens could directly yield 2.08 kg HDP from human food-crops. For dairy cows, the LUR was 2.10 when kept on sandy soil and 0.67 when kept on peat soil. The LUR for dairy cows on peat soil was lower compared to cows on sandy soil because land used to grow grass and grass silage for cows on peats was unsuitable for direct production of food-crops. A LUR <1.0 is considered efficient in terms of global food supply and implies that animals produce more HDP per square metre than crops.

Conclusions. Values <1.0 demonstrate that livestock produce HDP more efficiently than crops. Such livestock systems (with a LUR<1.0), therefore, do have a role in future food supply and, therefore, contribute to food security. Our LUR offers identification of livestock production systems that contribute to global food supply, i.e. systems that value land with low opportunity costs for arable production and/or co-products from crop cultivation or the food or energy industry.

1 Introduction

A growing and wealthier human population implies an increase in demand for their needs, such as housing, infrastructure, energy, and food, especially animal-source food (ASF). The current livestock sector already uses about 70 % of global agricultural land (FAO, 2009). The expected increase in demand for ASF, therefore, will further intensify global pressure on land. An increased pressure on land amplifies the risk of converting forests, wetlands or natural grasslands into agricultural land, resulting in emission of greenhouse gases and the loss of biodiversity and other important ecosystem services (Godfray et al., 2010; Foley et al., 2011). To limit land conversion, it is essential to e.g. improve land use efficiency of livestock systems.

It is generally acknowledged that increasing yields on existing land is key to improve land use efficiency in agriculture (Godfray et al., 2010; Tilman et al., 2011). Similarly, land use efficiency of livestock systems will improve with increasing yields of grazed pastures and feed crops per hectare. Land use efficiency of livestock systems, furthermore, can improve by increasing feed efficiency, i.e. the efficiency of converting feed into ASF (De Vries and De Boer, 2010). Besides increasing crop or animal productivity, land use efficiency of livestock systems improves also by increasing the efficiency along the entire food chain, from "field-to-fork", implying a reduction in, for example, grazing losses, losses while storing feed crops, or losses while consuming ASF.

It is increasingly recognized that, to achieve future food security, we might better not use highly productive croplands to produce feed for livestock. No matter how efficiently produced, direct consumption of cereals by humans is ecologically more efficient than consumption of ASF produced by animals fed with these cereals (Godfray et al., 2010; Foley et al., 2011). Improving land use efficiency of livestock systems, therefore, also implies feeding livestock mainly co-products from arable production or the food processing industry, that are not edible for humans; or grazing of livestock on "marginal land", i.e. land with low opportunity costs for arable production (Garnett, 2009; Eisler et al., 2014).

All above-described aspects are essential to improve land use efficiency of livestock systems to increase food supply and, therefore, contribute to food security. The urgent question remains, however, how land efficient are various livestock systems in terms of food supply? In this paper, we describe a novel method to calculate land use efficiency of livestock systems, which enables identification and improvement of systems that do have a role in future food supply. To illustrate our concept, we computed our novel method for three case systems in the Netherlands: production of ASF (eggs and meat) from laying hens and production of ASF from dairy cows (milk and meat from the dairy farm) on peat soil and on sandy soil.

To demonstrate the importance of our novel concept, we compared our findings with existing measures for land use efficiency. We, therefore, first describe in more detail how land use efficiency of livestock systems has developed and was measured to date.

2 Methods

2.1 Current drivers and existing measures of land use efficiency of livestock systems

In the past, improving land use efficiency of livestock systems was mainly driven by economic incentives and was directed at increasing productivity per hectare of land. Consequently, crop and animal productivity per hectare has increased enormously. For most cereal crops in the world, yields have increased almost linearly since 1960. Average US maize production, for example, increased 114 kg per hectare per year between 1960 and 2011 (Grassini et al., 2013). Similarly, feed efficiency of livestock has improved continuously. The feed conversion ratio (FCR) of broilers (i.e. kg feed used per kg of final body weight), for example, was estimated to reduce by 0.02 kg feed/kg broiler meat per year between 1960 and 2013 (Neeteson-van Nieuwenhoven et al., 2013).

The enduring focus on reducing FCR in livestock, however, also led to large amounts of human-edible plant products, like cereal grains, in livestock diets. Annually, about 1 billion tons of cereal grains are fed to livestock (Eisler et al., 2014). Direct consumption of these cereals by humans is more efficient in terms of global land use than consumption of products derived from livestock fed with these cereals because energy is lost during conversion from plant to animal product (Goodland, 1997). In a situation where land availability is no longer abundant, i.e. feeding a growing world population with a given amount of land, improving FCR in livestock might not necessarily imply improving global land use efficiency. To determine the role of livestock in terms of global food supply, we are in need of a measure that accounts for the competition for land between livestock and human.

One way to measure this competition for land is to compute human-edible protein and energy conversion ratios (Wilkinson, 2011; Dijkstra et al., 2013). These conversion ratios represent the amount of energy or protein in animal feed that is potentially edible for humans over the amount of energy or protein in that animal product that is edible for humans. Ratios above 1, such as for UK broilers, laying hens, pigs and some cattle, are unsustainable because animals produce less edible protein and/ or energy than they consume (Wilkinson, 2011). A ratio below 1, such as for UK milk production (Wilkinson, 2011), does not immediately imply efficient land use in terms of global food supply because these conversion ratios do not yet include the fact that, for example, grass fed to dairy cows can be produced on land suitable for the cultivation of human food-crops, or in other words, they do not include the opportunity costs of land for human food production.

The above-described conversion ratios originally focused on the efficiency of animals to convert feed or specifically human-inedible feed into animal products. Besides improving crop yield per hectare or the feed efficiency of animals, it is increasingly recognized that land use efficiency by livestock should be examined along the entire livestock supply chain. Over the last years, several studies were published that assessed land use by livestock along the entire supply chain (De Vries and De Boer, 2010), generally using life cycle assessment (LCA). At present, LCA is an internationally acknowledged method to quantify use of natural resources, such as land or fossil energy, during the entire life cycle of a product (Guinée et al., 2002). An LCA quantifies the land needed to produce 1 kg ASF and implicitly combines information about crop productivity (i.e. crop yield per ha) and animal productivity (i.e. feed efficiency along the chain, including breeding, rearing, and producing animals). Current LCA results show that production and utilization of feed are the dominant factors determining land use efficiency of livestock systems. Several LCA studies determined the land use efficiency of contrasting livestock products. They concluded that production of 1 kg of beef protein uses most land, followed by production of 1 kg of pork, chicken, egg, or milk protein (De Vries and De Boer, 2010; Eshel et al., 2014). Interpretation of current LCA results, however, is hindered because results do not include differences in consumption of human-edible products by livestock or differences in suitability of land used for feed production to directly cultivate food-crops, or in other words, they do not account for the competition between humans and animals for land. Grass-fed beef cattle, for example, generally consume less human-edible products than pigs or poultry and can value grassland that is less suitable for production of food-crops.

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Several LCA studies did propose a way to account for differences in quality of land (Ridoutt et al., 2012; Borucke et al., 2013). Net primary productivity of potential biomass (NPP_0, g C m^{-2} $year^{-1}$), for example, was used as proxy to account for differences in land quality (Ridoutt et al., 2012). According to this approach, land use of various agricultural products is assessed by multiplying each spatially differentiated area of land use by its net primary productivity divided by global average net primary productivity. Using net primary productivity as a proxy for land quality, however, does not yet include the fact that, for example, feed crops fed to dairy can be produced on land less suitable for the cultivation of human food-crops or, in other words, that livestock can produce human-edible protein from land with low opportunity costs for human food production.

Existing measures for efficiency of land use for livestock systems capture different aspects of the debate. The FCR focuses on the efficiency of animals to convert feed into animal products; protein and energy conversion ratios focus on the efficiency of animals to convert human-inedible feed into animal products; and an LCA focuses on the total amount of land required to produce 1 kg ASF and combines plant and animal productivity. None of these measures includes the opportunity costs of land for crop production. To address the contribution of livestock to increase food supply and, therefore, contribute to food security, we are in need of a measure for land use efficiency that accounts for plant productivity, efficiency of converting especially human-inedible feed into animal product and the opportunity cost of land for crop cultivation and has a life cycle perspective.

2.2 Novel measure for land use efficiency of livestock systems

Our measure of land use efficiency of livestock systems includes all above-mentioned aspects to determine the role of livestock in terms of food supply and is defined as the following land use ratio (LUR):

$$LUR = \frac{\sum_{i=1}^{n}\sum_{j=1}^{m}\left(LO_{ij} \times HDP_j\right)}{HDP\ of\ one\ kg\ ASF} \tag{1}$$

where LO_{ij} is the land area occupied for a year (m^2 year) to cultivate the amount of feed ingredient i (i=1,n) in country j (j=1,m) that is needed to produce 1 kg ASF, including breeding and rearing of young stock, and HDP_j is the maximum amount of human digestible protein (HDP) that can be produced per m^2 year by direct cultivation of food-crops in country j. The denominator contains the amount of HDP of 1 kg ASF.

To compute the LUR of 1 kg ASF from a specific livestock system, four steps are required. First, you quantify the land area occupied (LO_{ij}) to cultivate the amount of each feed ingredient (i=1,n) in the different countries of origin (j=1,m) that are needed to produce 1 kg ASF. Second, you assess the suitability of each land area occupied to directly grow human food-crops, using the crop suitability index (IIASA and FAO, 2012). Third, for each area of land suitable for direct cultivation of food-crops (LO_{ij}), you determine the maximum HDP_j from cultivation of food-crops by combining information about crop yield per hectare for each suitable crop, with its protein content and human digestibility. The amount of HDP that can be produced on all land required for feed production is summed and used as numerator. Fourth, you assess the amount of HDP in 1 kg ASF, which is the denominator.

LUR, therefore, represents the maximum amount of HDP derived from food-crops on all land used to cultivate feed required to produce 1 kg ASF over the amount of HDP in that 1 kg ASF. A ratio above 1 implies that the land required to produce this kilogramme ASF would yield more HDP if used directly to cultivate human food-crops, whereas a ratio below 1 implies that livestock production is the best way to produce HDP from that land.

The four steps of our concept will be further explained by computing the three case studies described below.

2.3 Computation of land use efficiency of case systems

To illustrate our concept, we computed our novel method for three case systems in the Netherlands: production of ASF from laying hens (eggs and meat) and production of ASF from dairy cows (milk and meat from the dairy farm) on peat soil and on sandy soil. We distinguished dairy farming on peat soil and sandy soil because of their difference in suitability to cultivate food-crops. Furthermore, we compared our novel method with currently available methods; FCR, the protein and energy conversion ratio, and LO_{LCA}, for protein and energy. The calculations for our novel method and the current methods are based on the same data.

Land use ratio

1. Quantify land area occupied to cultivate feed ingredients. We analysed the most common laying hen system in the Netherlands, i.e. a multi-tier barn system with brown hens. Our production system included the rearing and laying hen phase, whereas production of day-old hens and parent stock were excluded. Feed intake of rearing and laying hens was calculated based on technical data of KWIN-V (2013) (Table 1).

Table 1. Technical and economic data for Dutch egg production in a multi-tier barn system (KWIN-V, 2013).

Technical parameter	Value
Feed intake of hen in rearing phase (kg/rearing hen/round)	6
Mortality rate between 17 and 20 weeks (%)	0.3
Mortality rate from 20 weeks onwards (%)	10
Feed intake of hens from 17 weeks onwards (kg/hen/round)	48.8
Egg production (kg/hen/round)	21.2
Slaughter weight of laying hens (kg/hen)	1.8
Egg price (€/kg)	0.951
Slaughter price (€/kg)	0.18

Table 2. Composition of concentrate feed for rearing and laying hens (g/kg) (based on Dekker et al., (2011) and Gijsberts (2013a; 2013b; 2013c; 2013d)).

Feed ingredient	Rearing hen	Laying hen
Maize	411	539
Wheat	399	83
Soybean expeller	115	-
Soybean meal (0-45 CF; >480 CP)[a]	-	170
Sunflower seed expelled with hulls	75	-
Distillers dried grains and solubles (DDGS)	-	79
Fats/oils vegetable	-	21
Amino acids, minerals, enzymes and chalk[b]	-	108

[a] CF = crude fibre, CP = crude protein (g/kg)
[b] Components without associated agricultural land use

Rearing hens were assumed to consume only one type of concentrate feed. The composition of this concentrate feed was based on Dekker et al., (2011) and is reported in Table 2. Laying hens were fed a starter feed during the first 23 weeks, followed by a regular feed (personal communication, L. Start, Schothorst Feed Research, Lelystad, the Netherlands). The composition of starter and regular feed for laying hens was based on recent advices for commercial feed (Gijsberts, 2013a; Gijsberts, 2013b; Gijsberts, 2013c; Gijsberts, 2013d). The weighted average of the starter and regular feed of laying hens is presented in Table 2.

For each feed ingredient, the country of origin and yields per hectare were based on a database called 'Feedprint', Feedprint (based on currently available literature) provides information on the environmental impact of feed ingredients used in the Netherlands (Vellinga et al., 2013). Given the exact amount of feed ingredients consumed, and their yields per hectare, we quantified the area occupied to cultivate all feed ingredients. In case of a multiple-output situation, land use was allocated to the various outputs based on their relative economic value (i.e. economic allocation). Crop residues, such as citrus and beet pulp, maize gluten meal, and straw, were assumed to have an economic value of zero (Vellinga et al., 2013).

For dairy farming, we selected dairy production systems in the Netherlands with >90% sandy soil or >90% peat soil. Technical data required to determine all land used to cultivate feed required to produce 1 kg ASF from this system were based on the Dutch Farm Accountancy Data Network (FADN, 2014) and are shown in Tables 3 and 4. Feed intake of the whole dairy herd, including young stock, was based on average technical data between 2010 and 2012 Table 3.

Table 3. Average technical and economic data for Dutch dairy production system on peat (n = 23) or sand (n = 100) (FADN, 2014).

Technical parameter	Dairy Peat	Dairy Sand
Number of milking cows	90	94
Number of young stock < 1 year	27	35
Number of young stock > 1 year	29	35
Milk production per cow (kg/year)	6353	8114
Milk production per ha (kg/year)	10623	15118
Total milk production per farm (kg/year)	571912	761795
Fat in milk (%)	4.38	4.41
Protein in milk (%)	3.51	3.54
Total sold meat (kg live weight/year)	12060	17368
Feed intake parameters		
Grass intake via grazing (GJ NE_L[a])	630.7	321.9
Grass intake via silage (GJ NE_L)	2043.3	1704.1
Maize silage[b] (GJ NE_L)	540.5	1764.6
Concentrate feed (GJ NE_L)	1049.3	1187.7
Wet co-products (GJ NE_L)	328.9	206.4
Milk products for young stock (GJ NE_L)	18.1	20.5
Concentrate feed total N (kg)	4664	6436
Wet co-products total N (kg)	826	953
Crop yields		
Grass yield (GJ NE_L /ha)	67.9	70.2
Grass yield (kg DM/ha)	9157	9544
Grass yield (kg N/ha)	255	253

[a] NE_L = Net energy for lactation
[b] Including a small amount of feed reported as 'other' (27.9 GJ NE_L for Dairy Peat; 75.8 GJ NE_L for Dairy Sand)

Table 4. Feeding and conservation losses (in %) for roughages and wet co-products (RIVM, 2013).

	Feeding losses	Conservation losses		
		DM	NE_L[a]	N
Grass silage	5	10	15	3
Maize silage	5	-[b]	-	-
Concentrates / milk products (%)	2	0	0	0
Wet co-products (%)	2	-	6	-

[a] NE_L = Net energy for lactation
[b] Not used in calculations, as nutritional value of purchased products was derived from Feedprint

Table 3 shows average technical and economic data for Dutch dairy production system on peat or sand. To quantify the land occupied for the cultivation of a feed ingredient, we corrected these data for feeding and conservation losses (Table 4). Feeding and conservation losses of grazing animals were assumed negligible (RIVM, 2013). Furthermore, we assumed all maize silage to be purchased because no data were available to exactly determine the production of on-farm maize silage.

Purchased concentrates were assumed to be 70.5% protein-rich (19.6% crude protein) and 29.5% very protein-rich (30.4% crude protein) for dairy cows on sandy soil, and 16.2% standard (14.1% crude protein) and 83.8% protein-rich for dairy cows on peat soil (Table 5). Table 5 shows the composition of each type of concentrate. The amount of both concentrate types purchased was computed by combining information from the total amount of crude protein and energy in purchased concentrates (Table 3) and the crude protein-to-energy ratio of both types of concentrates (Vellinga et al., 2013). Given the amount of concentrates consumed, and their related average yields per hectare, we quantified the area occupied to cultivate all feed ingredients (Vellinga et al., 2013).

Similarly, the amount of purchased wet co-products, brewers grain and sugar beet pulp was computed by combining information from the total amount of crude protein and energy in purchased wet co-products (Table 3) and the crude protein-to-energy ratio of both products. Given the amount of feed ingredients consumed, and their yields per hectare, we quantified the area occupied to cultivate all feed ingredients. For on-farm feed production, yields per hectare were obtained from the Dutch Farm Accountancy Data Network (FADN) (2014); Table 3). For crop production outside the farm, country of origin and yields per hectare were obtained from Feedprint (Vellinga et al., 2013).

2. Determine the suitability of land to directly cultivate human food-crops. For each area of land identified for feed production, we determined its suitability to directly grow food-crops. On each area of land suitable to grow food-crops, different crops can be cultivated. We focused on the five major food-crops as the yield of those crops are high and, therefore, the amount of protein per hectare is also high. The suitability for the five major food-crops, i.e. wheat (wetland and indica dry land), maize, potatoes (white and sweet) and soybeans, at a specific location was assessed based on the Global Agro-Ecological Zones (GAEZ) database (IIASA and FAO, 2012). This database classifies crop suitability in eight groups (varying from very high to not suitable), by quantifying to what extent soil (e.g. pH, soil water holding capacity) and climatic conditions (e.g. wet day frequency, sunshine, temperature) match crop requirements, under defined input and management circumstances. We assessed crop suitability for current cultivated land in a situation of high input levels, optimal water supply,

and baseline climate conditions (1961–1990). If the suitability of the crop was good, high, or very high (suitability index >55), the land was considered suitable to cultivate that specific crop.

Table 5. Composition (g/kg) of three types of concentrate feed for dairy cattle (standard, protein-rich (Rich) and extra protein-rich (Extra)) (Vellinga et al., 2013).

Feed ingredient	Standard	Rich	Extra
Citrus pulp dried	250	156	65
Coconut expeller CFAT > 100[a]	100	100	100
Maize gluten feed CP 200-230[b]	185	370	-
Milk powder whole	8	8	8
Palm kernel expeller CF 0-180[c]	150	150	150
Rapeseed expeller	-	50	-
Rapeseed extruded CP 0-380	-	-	126
Soybean hulls CF 320-360	150	-	-
Soybean meal CF 45-70; CP 0-450	-	-	320
Soybean meal Mervobest	-	76	140
Sugarcane molasses SUG>475[d]	30	30	30
Sunflower seed expelled with hulls	-	-	-
Triticale	13	-	-
Vinasse sugarbeet CP 0-250	40	40	40
Wheat middlings	73	8	-
Others, like salt, chalk[e]	1	14	22

[a] CFAT = crude fat (g/kg)
[b] CP = crude protein (g/kg)
[c] CF = crude fibre (g/kg)
[d] SUG = sugar (g/kg)
[e] Components without associated agricultural land use

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Table 6. Country average yields (kg/ha) of five major food-crops for the year 2011 (FAOstat; http://faostat.fao.org). An empty cell implies a country was considered unsuitable to cultivate that crop (i.e. suitability index <55).

Country	maize	potatoes (sweet)	Potatoes (white)	rice (dry)	rice (wet)	soybeans	wheat
Argentina	6350	15083	30383	-	6790	2605	3136
Australia	5739	24546	35089	-	9544	1714	2030
Belgium	-	-	50141	-	-	-	8405
Brazil	-	-	12427	4896	4896	3121	-
China	5748	-	16281	-	6686	1836	-
France	9973	-	46899	-	-	2947	6527
Germany	-	-	45613	-	-	-	7019
India	2498	9246	-	-	3591	1200	2989
Indonesia	-	12326	-	4980	4980	1359	-
Malaysia	-	10655	-	3898	3898	-	-
Netherlands	-	-	46055	-	-	-	7781
Philippines	2740	4979	-	3678	3678	1323	-
Pakistan	-	-	-	-	-	-	-
Pakistan	-	-	-	-	3720	-	-
Sudan	1351	22688	-	-	-	-	3353
Ukraine	6445	-	-	-	-	-	7749
United Kingdom	-	-	41884	-	7921	2820	2942
United States	9237	-	44714	-	6790	2605	3136

Table 7. Dry matter (DM), protein and human-digestible (HD) energy contents of products and human digestibility value of protein.

Product	Product code[a]	DM (kg DM/kg product)	HD energy (MJ/kg DM)	Protein (g/kg DM)	Protein digestibility[c] (%)
Chicken egg	01123	0.239	25.1	526.6	97
Chicken meat	05001	0.337	26.5	544.6	94
Cow milk (sand)	01078	0.123	27.0[b]	287.8[b]	95
Cow milk (peat)	01078	0.123	26.9[b]	285.4[b]	95
Beef	13002	0.418	27.9	418.3	94
Maize	20014	0.896	17.0	105.1	85
Potatoes sweet	11507	0.227	15.8	69.1	76
Potatoes white	11354	0.184	15.7	91.2	80
Rice	20052	0.867	17.3	75.0	89
Soybeans	16111	0.915	20.4	399.0	78
Wheat	20074	0.904	15.8	125.1	87

[a] Product code in USDA database (USDA, 2013) used to select values for DM, protein and HD energy
[b] Case specific data were used (see Table 3)
[c] Source: Gilani et al., (2005), except for potatoes (white (Kies and Fox, 1972; Eppendorfer et al., 1979; Khan et al., 1992; Gahlawat and Sehgal, 1998); sweet (Ravindran et al., 1995)

3. Calculation of HDP production from all land suitable for crop production (numerator). Human-digestible protein production from the five selected crops was calculated from their respective yields (Table 6), multiplied by their protein content and digestibility (Table 7). For on-farm land used for grass production, crop yields per hectare were assumed to be soil specific. Peat soil were assumed unsuitable for the cultivation of any of the five major food-crops, whereas sandy soil were assumed suitable for cultivation of white potatoes (i.e. 56000 kg/ha) and wheat (7300 kg/ha) (KWIN-AGV, 2012). For off-farm crop production, country-average yield data from FAOstat were used (Table 6), as information about exact location and, consequently, soil type were missing. Subsequently, the highest HDP for each area of land was chosen and summed across all land areas required to produce 1 kg of ASF. This sum of HDP was used as numerator of our land use ratio. Because ASF contributes not only to the protein but also to the energy demand of humans, we also computed our land use ratio from an energy perspective, implying that HDP was replaced by human-digestible energy. For human-digestible energy, values of the energy content (Table 7) were directly derived from a USDA database (USDA, 2013). For chicken meat, we assumed that 56% of live weight was edible and for beef, this was 43% (De Vries and De Boer, 2010).

4. Calculation of HDP of 1 kg of animal-source food (denominator). The amount of HDP in 1 kg of ASF was computed by multiplying with its protein content and its protein digestibility for humans. The amount of human-digestible energy in 1 kg of ASF was computed by multiplying with its energy content for humans.

Assessing existing measures of land use efficiency of livestock systems

To demonstrate our concept, we compared LUR values with existing measures of land use efficiency. To allow an accurate comparison, we computed existing measures (FCR, protein and energy conversion ratios, and LO_{LCA}) for each case system using the same data as we used to compute LUR.

The feed conversion ratio (FCR) was defined as the amount of DM in feed supplied to the producing animal over the kilogramme of main output of that animal (kg egg and kg fat-protein-corrected milk) (CVB, 2012). A higher FCR indicates a lower efficiency of converting feed into animal product and implies a lower efficiency of land use. Feed intake was already assessed in the first step of our novel approach. Feed intake data as presented in Table 3, however, were computed for the entire herd. To correct for feed intake of young stock in the computation of feed, energy and protein conversion ratios, we assumed a total intake from birth until first lactation of 31 GJ NE_L per animal (NE_L = net energy for lactation), of which

19% consisted of concentrate feed and milk products (CVB, 2012; Dijkstra et al., 2013). All milk products as presented in Table 3 were assumed to be consumed by young stock. Average concentrate feed and roughage composition was assumed to be similar for young stock and producing cows per case system. No wet co-products were assigned to young stock. Based on those assumptions, we calculated a total feed intake of dairy cows on sandy soil of 246 GJ NE_L grass grazing, 1300 GJ NE_L grass silage, 1346 GJ NE_L maize silage, 1001 GJ NE_L concentrate feed and 206 GJ NE_L wet co-products and dairy on peat soil of 491 GJ NE_L grass grazing, 1590 GJ NE_L grass silage, 421 GJ NE_L maize silage, 902 GJ NE_L concentrate feed and 329 GJ NE_L wet co-products. The DM content of feed ingredients was taken from Feedprint (Vellinga et al., 2013), whereas the DM content of on-farm grass was taken from the FADN (2014) (Table 3).

The protein conversion ratios were defined as the ratio of crude protein in animal feed directly edible for humans over kilogramme protein in eggs or milk. Similarly, the energy conversion ratio was defined as the ratio of gross energy in animal feed directly edible for humans over kilogramme gross energy in eggs or milk. A ratio above 1 implies that an animal produces less edible protein than it consumes and appears inefficient from a land-use perspective. The human-edibility of feed products was taken from the literature (Wilkinson, 2011), whereas ASF was considered to be fully human-edible. We used average nutrient composition values for eggs (USDA, 2013), whereas for milk, we used case-specific protein contents (Table 3).

Besides examining feed and protein conversion ratios at animal level, we also computed land use associated with feed production along the life cycle of 1 kg HDP from laying hens or dairy cows, including rearing of young stock. Land area occupied to cultivate feed ingredients along the chain was assessed already in the first step of our novel approach (see previous paragraph). Subsequently, it was allocated to the main product of the livestock system (i.e. egg or milk) based on economic allocation and expressed per kilogramme of human-digestible protein or per kilogramme of human-digestible energy.

3 Results

We first present LUR values from a ‘protein’ perspective, as livestock products contribute especially to the protein demand of humans (Galloway et al., 2007; De Vries and De Boer, 2010) and compare those with results from existing measures of land use efficiency. Second, we present and compare results from an energy perspective.

3.1 Results of land use ratio (protein perspective)

The LUR for the case of laying hens equalled 2.08. A LUR of 2.08 implies that the land required to produce 1 kg HDP from laying hens could directly yield 2.08 kg HDP from human food-crops. The structure of the computation of the LUR is depicted in Figure 1. To produce 1 kg of fresh egg and its associated production of 0.068 kg chicken meat, we needed 2.30 kg of feed for laying hens and 0.284 kg of feed for rearing hens. The main feed ingredients of laying hen feed were maize (54%), soybean meal (17%), and wheat (8%). Half of this maize was assumed to originate from Germany and the other half from France. With a yield of 8788 kg per hectare, production of 0.62 kg maize in Germany required 0.71 m^2 year. This 0.71 m^2 year could have been used directly to produce human food-crops and could yield maximally 0.049 kg HDP ((7019 kg wheat per ha×0.904 DM per kg wheat×125.1 g protein per kg DM wheat × 87% digestibility / 10000) × 0.71 m^2 year). In total, the land used to produce 1 kg of eggs and associated chicken meat could have yielded directly 0.27 kg of HDP from human food-crops. One kg of eggs contains 0.12 kg HDP, whereas 0.07 kg of chicken meat contains 0.01 kg HDP. The LUR of eggs, therefore, equalled 0.27/0.13=2.08.

Similarly, we determined a LUR of 2.10 for dairy cows on sandy soil and 0.67 for dairy cows on peat soil. The LUR of dairy cows on sand was similar to the LUR of hens, despite the fact that compared with the diet of laying hens, the diet of dairy cows contained less products that humans could consume directly (i.e. 72% of crude protein in diets of laying hens was human-edible compared to 16% for dairy cows on sand and 9% for dairy cows on peat). The land used to produce feed ingredients for laying hens and dairy cows on sandy soil, however, appeared to have about the same potential to directly produce HDP by food-crops. This was not the case for dairy cows on peat soil. The land used to grow grass and grass silage for these dairy cows was assumed unsuitable for direct production of food-crops, overall resulting in a LUR of 0.67. This LUR implies that the land required to produce 1 kg HDP from dairy cows on peat soil could only yield 0.67 kg of HDP from human food-crops directly. A LUR <1.0, therefore, is considered efficient in terms of global food supply and implies that animals produce more HDP per square metre than crops. Values <1.0 demonstrate that livestock do contribute to food supply and, therefore, food security. Our LUR offers identification of livestock production systems that use land efficiently in terms of food supply. Land-efficient livestock systems typically value land with low opportunity costs for arable production (e.g. peat soil or wet grasslands) and/or co-products from crop cultivation or the food or energy industry (e.g. beet pulp).

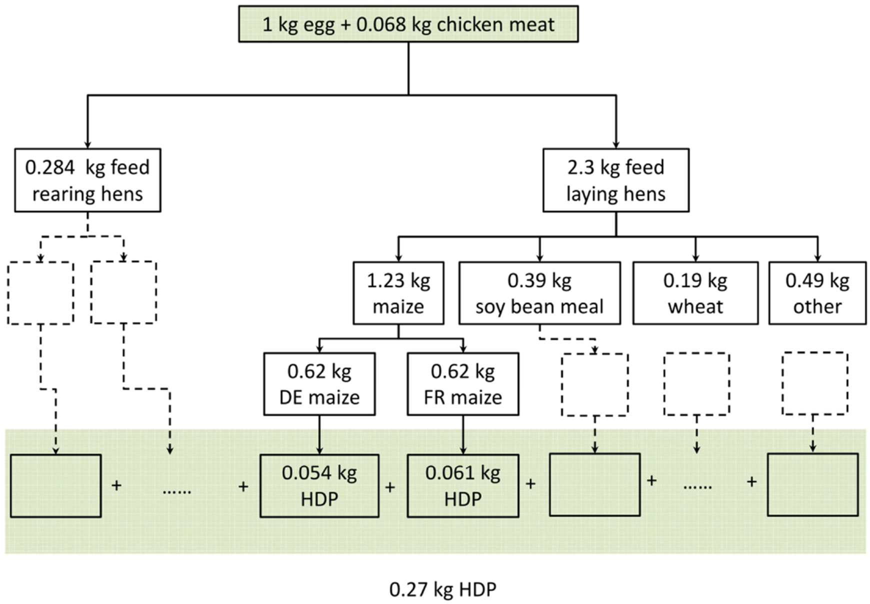

Figure 1. Illustration of our concept of land use ratio for the case system of laying hens, assuming a production of 1 kg of eggs and an associated production of 0.076 kg of chicken meat (DE Germany, FR France, HDP human-digestible protein).

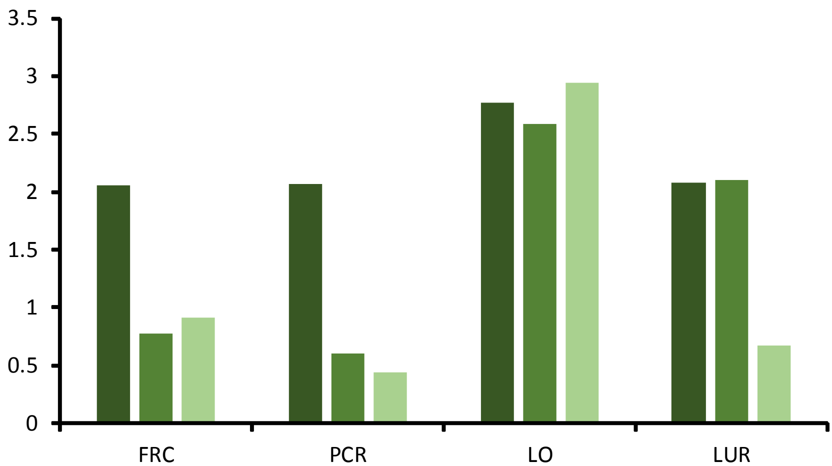

Figure 2. Feed conversion ratios (FCR) in kg dm/kg product (i.e. eggs or milk), protein conversion ratios (PCR) in kg human-edible protein/kg human-edible product, life cycle assessments of land occupation (LO) in 10m² year/kg human-digestible protein product and our newly developed land use ratio (LUR) in kg human digestible protein in crops/human-digestible protein in animal source food. All methods are applied for three case systems in the Netherlands, laying hens (dark green), dairy cows on sand (middle green) and dairy cows on peat (light green).

3.2 Results of existing measures of land use efficiency of livestock systems

The FCR for producing eggs is roughly twice as high as the production of milk on peat (Figure 2), which is in line with a UK case study (Wilkinson, 2011). The FCR of dairy cows on sandy soil, however, was about 15% lower than the FCR of dairy cows on peat soil. Differences in FCR are determined mainly by differences in annual milk production per cow (Dijkstra et al., 2013). Annual milk production per cow indeed was comparable for Dutch cows on peat (i.e. 6350 kg) and cows in the UK case study (Dijkstra et al., 2013) (6500 kg) but was higher for dairy cows on sand (8114 kg). Based on this definition of FCR, we would conclude that dairy cows are more efficient than laying hens, whereas dairy cows on sand are most efficient. Relative to eggs, however, milk has a lower DM content (i.e. milk = 12.3% DM; eggs = 23.9% DM). When we express FCR as kg DM in feed over kg DM in product, differences among FCRs between milk production and egg production are less pronounced (FCR_{egg} = 8.6; $FCR_{milk\ sand}$ = 6.3; $FCR_{milk\ peat}$ = 7.4), which is in agreement with Galloway et al. (2007).

The protein conversion ratios of laying hens are higher compared to dairy cows, which is in the range with results in literature (Wilkinson, 2011; Dijkstra et al., 2013). Our laying hen system shows a protein conversion ratios of about 2. In terms of global food supply, therefore, the existing way of egg production is not land efficient. A target ratio below 1 may be possible by replacing, for example, cereals or soybean meal (both have a high proportion of edible protein) with waste-fed insects or with co-products from the food or energy industry with a low economic value. Our dairy systems show a protein conversion ratios <1.0, clearly demonstrating the ability of ruminants to turn human-inedible feed ingredients into human-edible product. The protein conversion ratios of dairy cows on peat was lower (0.44) than of cows on sand (0.60) because cows on peat consume relatively more grass. Grass has a relatively high protein content. Concentrates fed to dairy cows on peat, therefore, have a lower protein content than concentrates fed to cows on sand. In contrast to grass, some ingredients in concentrates are human-edible. The difference in protein conversion ratios between cows on peat and sand, therefore, is explained by the difference in protein content of concentrates fed to cows on peat and sand. Based on the protein conversion ratio results, we would conclude that dairy cows are more efficient than laying hens, whereas dairy cows on peat are most efficient.

Land use for production of 1 kg HDP from laying hens required slightly more compared to the production of 1 kg HDP from dairy cows on sandy soil but slightly less compared to the production of 1 kg HDP dairy cows on peat soil. Based on these Dutch case studies, therefore, we would not conclude that dairy production is more efficient than egg production. Moreover, production of milk on peat soil appears least efficient.

Table 8. Energy conversion ratios (ECR), life cycle assessments of land occupation based on energy ($LO_{LCA,energy}$) and our newly developed land use ratio based on energy (LUR_{energy}) for three case systems.

Case systems	ECR MJ GE HE[a] feed/ MJ GE HE product[b]	$LO_{LCA,energy}$ m^2.yr/ MJ HDE[c] product	LUR_{energy} MJ HDE in crop/ MJ HDE in ASF[d]
Laying hens	3.91	0.56	6.39
Dairy cows on peat	0.39	0.33	1.22
Dairy cows on sand	0.38	0.29	4.35

[a] GE HE = gross energy human-edible
[b] product = egg or milk
[c] HDE = human-digestible energy
[d] ASF (animal-source food, including meat)

3.3 Results from an 'energy' perspective

Besides protein, ASF in many parts of the world also contributes to the energy demand of humans. For existing measures of land use efficiency, the main conclusions from the comparison among livestock systems presented in this study are valid also when presented from the 'energy' perspective, albeit slightly numerically modified (Table 8).

Using human-digestible energy instead of HDP in our computation of LUR, however, yielded different results: 6.39 for laying hens, 1.22 for dairy cows on peat soil, and 4.35 for dairy cows on sandy soil. These results demonstrate that none of our Dutch case systems produced human-digestible energy more efficiently than crops and support earlier findings that plants produce energy more efficiently than protein (Penning De Vries et al., 1974), whereas for livestock, this is reversed (Phuong et al., 2013). From the perspective of food supply, therefore, the main role of livestock in a human diet is provision of protein.

4 General discussion

The fact that livestock especially contribute to the protein demand of humans justifies our choice for HDP in the LUR. Protein digestibility was taken into account to correct for differences in protein quality between plant and animal products, whereas differences in essential amino acid content between plants and animals were not accounted for. Besides protein, ASF has other nutritional qualities, such as the provision of iron and vitamin B_{12}. In principle, we could extend our LUR not only to consider human-digestible protein but also to include, for example, available iron, calcium, essential amino acids or vitamins. This, however, would require an index for nutritional quality of a food item. Such indices have been developed for individual food items and are referred to as nutrient density scores. These density scores relate the nutrient content of 100 g or 100 kcal of a product to the daily recommended intake and average the values of different nutrients into one final score (Drewnowski and Fulgoni, 2014). Despite a low nutrient density score, however, an individual food item can be valuable at the dietary level because of its richness in one very scarce nutrient. We believe, therefore, that the full range of nutritional needs should be met at the level of the entire human diet.

By applying the LUR method, it is possible to increase land use efficiency. However, to identify the contribution of livestock to future sustainable diets, one should also assess the contribution of livestock systems to global warming, acidification, eutrophication, water use, biodiversity and other environmental impacts. For example, a ruminant system on marginal grassland with a LUR <1 might have a relatively high global warming potential as feeding fibrous diets increases the production of enteric methane but could also contribute to the preservation and enhancement of biodiversity, and the conservation of cultural landscapes when grasslands are managed well. We demonstrated the LUR for three case systems: laying hens, dairy cows on sandy soil, and dairy cows on peat soil. These case systems represented existing livestock systems and were deliberately chosen because literature showed that they had comparable land use requirements from a life cycle perspective, while they differed in the percentage of human-edible feed in diets of animals and in opportunity costs of land for crop cultivation (sandy versus peat soil). Moreover, computation of LUR requires global, high-resolution inventory data, which were partly available for these existing systems. Despite the data availability, the LUR estimate of our case systems could have been refined further if the exact production location and associated yields of all purchased feed ingredients would have been known. Such detailed information about feed ingredients, however, is generally absent.

Our concept is applicable to a large variety of livestock systems and is of paramount importance in the debate about the role of livestock in global food supply. Several, especially LCA-based studies recommended to switch from an animal-based to a plant-based human diet, whereas others advice to substitute beef by pork or chicken to minimize land use in an animal-based diet (Stehfest et al., 2009; Meier and Christen, 2012; Eshel et al., 2014). The above-mentioned studies, however, do not account for differences in the suitability of land to directly produce food-crops. Our results clearly demonstrate that ruminant systems that value land with low opportunity costs for arable production can produce HDP more efficiently than crops and, therefore, do have a role in future food supply. In a situation of land scarcity, therefore, a plant-based diet is not more land efficient than a diet including animal-source food from, for example, ruminants grazing on land less suitable for crop production. Analogously, simply substituting beef by pork or chicken does not automatically imply improving efficiency of land use in terms of food supply. Beef and/or milk produced from grass on peat soil only would even result in a LUR of zero.

Our LUR enables identification of land-efficient livestock systems and allows further improvement of systems regarding efficiency of land use to contribute to future food supply. Land-efficient livestock systems (i.e. LUR <1.0) typically value land with low opportunity costs for arable production (e.g. beef or dairy cattle grazing 'marginal land') and/or co-products from food or bio-energy production (e.g. pigs eating beet-pulp or rapeseed meal). The amount of ASF that can be produced from 'marginal land' and co-products might not be sufficient to feed each human being a Western European or American diet. We acknowledge, therefore, that improving land use efficiency of livestock systems implies a more modest consumption of ASF in affluent countries. In countries where dietary diversity is limited and malnutrition levels are high, however, an increase in consumption of ASF is legitimate. A modest consumption of ASF is required also to temper environmental impacts of current and expected future demands of ASF. To use land efficiently, therefore, we should aim at increasing livestock productivity while maintaining a LUR <1.0.

5 Conclusions

Our results demonstrate that existing measures for efficiency of land use for livestock systems give insight into different aspects of the debate about the contribution of livestock to food supply and, therefore, food security. Conversion ratios are used to gain insight into the ability of animals to convert feed, or more specifically human-inedible feed, into animal products. Results show that improving the conversion of human-inedible feed into animal product improves land efficiency only if feed is produced on land with low opportunity costs for arable production (i.e. protein conversion ratios is lower for dairy cows than for laying hens, whereas LUR is similar for dairy cows on sandy soil and for laying hens). LCA results are used to gain insight into the land required to produce 1 kg ASF along the entire chain. Land use per kilogramme ASF reduces by increasing crop yield per hectare, reducing the feed conversion ratio and increasing the reproductive performance of animals. Land requirements per kilogramme ASF for cows on sand indeed are lower than for cows on peat, mainly because of the reduced FCR. This reduced FCR, however, also implied a diet with more human-edible plant products, i.e. the protein conversion ratios of cows on sand indeed was higher than of cows on peat. Improving LCA results of land use, therefore, might indirectly increase the amount of human-edible feed in diets of livestock and, as such, reduce the efficiency of land use in terms of food supply. None of the above-mentioned measures accounts for the opportunity costs of land to cultivate human food-crops. Our LUR includes all aspects of importance to determine the role of livestock for future food supply, and, therefore, yields a more complete insight into land use efficiency for livestock systems. Results demonstrated that ruminant systems that value land with low opportunity costs for arable production can produce HDP more efficiently than crops and, therefore, do have a role in future food supply. Values <1.0 demonstrate that livestock do contribute to food supply and, therefore, to food security. Our LUR offers identification of livestock production systems that use land efficiently in terms of food supply. Land-efficient livestock systems typically value land with low opportunity costs for arable production (e.g. peat soil or wet grasslands) and/or co-products from crop cultivation or the food or energy industry (e.g. beet pulp).

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Chapter 8

Opinion paper: The role of livestock in a sustainable diet: a land-use perspective

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In 2000, the Food and Agricultural Organisation (FAO) projected that global demand for animal source food (ASF) would double by 2050 (Alexandratos and Bruinsma, 2012). Although these projections were revised slightly during recent years, they form the basis of many scientific and policy documents related to livestock production. Those projections, however, are based on global trends for a growing population and increasing incomes and urbanization, but not based on ensuring global nutrition security in a sustainable way. Currently, the world's livestock sector adds to the total anthropogenic emissions of greenhouse gases and competes for scarce resources, such as land, water, and fossil-energy. Without changes to reduce the environmental impact, concerns about the environment will only increase further.

We asked ourselves, how and why livestock production is essential and what would be the proportion of ASF in human diets to ensure nutrition security in a sustainable way? As land is a strict limitation of nutrition security, we took a land-use perspective, irrespective of socio-economic or technical constraints. In 2012, about 4.92 billion ha was used for agriculture, of which about 70% was used for livestock production, mainly for pasture and production of feed crops (FAO stat). Of the 4.92 billion ha of agricultural land about 1.56 billion ha is used for crop production. Assuming 9.7 billion people in 2050, then about 0.16 ha of cropland is available per person. Production of a vegan diet, for example, requires about 0.14 ha per person. Expanding the area for crop production will lead to loss of grazing areas or deforestation in the tropics, for example, resulting in loss of biodiversity and increased carbon emissions. High productive croplands, therefore, must be used to produce human food instead of livestock feed. No matter how efficiently food is produced, direct consumption of cereals by humans is more efficient ecologically than consumption of livestock fed these cereals.

Should we shift, therefore, to vegan diets? Not necessarily! Grass-based ruminant systems on marginal land, that is, land not suitable for crop production, produce human digestible protein more efficiently than food-crops (Van Zanten et al., 2015c). Furthermore, compared with a vegan diet, consumption of a small amount of ASF reduced land use per person when livestock were mainly fed with co-products (Van Kernebeek et al., 2015).

In addition to biomass from marginal land and co-products, livestock can also upgrade two other biomass streams that humans do not currently consume: crop residues and food-waste. Using crop residues as livestock feed, however, can lead to depletion of soil organic carbon, and, therefore, should be left on the field. To be safe, we assumed all crop residues are left on the field. We focus, therefore, on the potential of livestock to convert co-products from human food, food-waste, and biomass from marginal land, referred to as 'leftover streams,' into high-quality ASF. Livestock that eat these leftover streams do not compete with humans for cropland, and, therefore, contribute to sustainable nutrition security. By feeding only leftover streams to livestock, the number of humans fed per hectare is maximized. How much ASF can we consume, however, when we want to avoid feed-food competition by feeding only leftover streams to livestock? To illustrate that we can produce a sufficient amount of ASF, we calculated amount of ASF produced from co-products and food-waste, and amount of ASF produced from 100% grass-based systems.

Amount of ASF produced from co-products and food-waste depends on availability, which depends on consumption patterns of humans. If the 1.56 billion ha of cropland is used for human food production only, people consume a vegan diet because no cropland is used for feed production. Consumption of a vegan diet requires annual production of about 129 kg co-products per person (see Appendix A.1, A.2, and A.3). We chose those food ingredients in a vegan diet, whose co-products had a high nutritional value for livestock. We assumed, for example, that oil production originates from soy cultivation resulting in soybean meal. Soybean meal compared with other co-products from oil processing, for example, sunflower meal, has a high nutritional value for livestock. This assumption not only has an impact on the final protein production from pork, but also demonstrates the importance of optimizing crop production based on food and feed use.

During production and consumption of food, furthermore, about one-third is wasted (Gustavsson et al., 2011). Reducing food-waste has greater environmental benefits than feeding food-waste to livestock. We assume nevertheless, that 10% of our food will be wasted, resulting in 46 kg annual food-waste per person, which can be used as livestock feed (see Appendix A.3). Most wasted food and some co-products have high digestibility and nutritional value for ruminants (e.g. cattle) and monogastrics (e.g. pigs). Using products with high digestibility, however, is more desirable for monogastrics than for ruminants, because enteric fermentation is lower for monogastrics then for ruminants. Through use of co-products and food-waste from an average vegan diet, we are able to fatten annually about 0.42 growing-pig per person. Based on an average final BW of pigs of 116 kg, about 71 g pork containing 14 g protein per person per day can be consumed (see Appendix A.4 and A.5 for calculation).

In addition to food-waste and co-products, biomass from permanent meadows and pastures can be valued by livestock, more specifically by ruminants, from production of milk or meat or both. Some of permanent meadows and pastures are on marginal land because of rainfall, temperature or terrain limitations. There are about 1.6 billion ha of marginal land, based on Global Agro-Ecological Zone (GAEZ) (Alexandratos and Bruinsma, 2012). If we use marginal land for production of ASF, we can produce daily about 3 g of protein per person. Production of 3 g protein assumes that we have 100% grass-based systems, livestock density of 0.5

tropical livestock unit (TLU) per ha, and protein production of 14 kg per TLU per year (see Appendix A.6). On a global scale, producing biomass from marginal land appears to be of less importance than producing protein from co-products and food-waste. On a local scale, however, marginal land can play an important role, for example, in food security in smallholder systems in developing countries.

Only part of the total area of permanent meadows and pastures is on marginal land. If we use the total area of permanent meadows and pastures for production of ASF, we can produce daily about 7 g of protein per person (see Appendix A.6). Based on GAEZ, about 1.4 billion ha currently used for grazing has potential for crop production. The purpose of this ‘grazing land,’ however, is debatable. If 0.16 ha per person is insufficient to produce enough food to provide the world population a vegan diet, part of 1.4 billion ha currently used for grazing must be transformed to cropland for food production. Of the 0.16 ha per person, about 0.14 ha is needed for the production of a vegan diet, which accounts for only 10% food-waste and does not include, for example, the production of cotton for clothes (see Appendix A.6). The area needed for crop production, furthermore, depends also on future developments of crop yields.

If, however, 0.16 ha per person is sufficient to produce enough food and other human needs then we have three options for the grazing land. First, we can continue the current practice of maintaining grazing systems, partly supplemented with concentrates. Second, we can increase production of ASF per hectare by transitioning from grazing systems to mixed crop-livestock systems. Third, we can use the land for purposes other than food production, for example, nature conservation, bio-energy or both. The amount of protein that can be produced from the total area of grazing land while avoiding feed-food competition, therefore, depends on the number of people to be nourished and production system chosen. In any case, a production of 7 g of protein from ASF per person per day seems to be feasible.

To sum up, in total about 21 g of protein from ASF can be produced person per day. The recommended intake of protein is about 60 g per person per day, from which about a third is recommended to be from AFS. These 21 g from AFS is produced without competing with food-crops for arable land. We can satisfy, therefore, the daily recommended intake of protein of person while avoiding competition for land between feed and food production.

What does this conclusion imply for the current situation? In practice, co-products and biomass from marginal land are already used. Co-products are used in animal diets and parts of marginal land are used by grass-based systems, sometimes so intensively that grasslands are degraded by overgrazing. Food-waste is the main unused source of leftover streams, which is of interest because of its high nutritional value. Use of food-waste is prohibited in many countries, including European countries, because of problems of health safety issues related to, for example, foot and mouth disease, African swine fever and Bovine spongi-form encephalopathy. Besides health safety issues, we should consider also alternative applications of food-waste. In the Netherlands, for example, food-waste is currently used for anaerobic digestion to produce bio-energy, which is used to replace fossil-energy. It is more effective, however, to replace fossil-energy with wind or solar energy than with bio-energy and to use food-waste instead for livestock feed (Van Zanten et al., 2015a). The FAO also recognizes the importance of using food-waste and, therefore, started an e-conference: 'Utilization of food loss and waste as well as non-food parts as livestock feed.'

To avoid feed-food competition, therefore, future innovations should focus on shifting diets, and on adapting livestock systems to use co-products, food-waste, and biomass from marginal land in livestock feed. To avoid feed-food competition consumption patterns in mainly developed countries must change. The average protein intake is, for example, about 61 g of animal protein per person per day in the EU. To reduce the consumption of ASF a transition route is, therefore, required. Furthermore, innovations are needed to overcome food safety problems and technical concerns related to collecting the leftover streams. Livestock systems should change their focus, therefore, from increasing productivity per animal toward increasing protein production for humans per hectare, which means making optimal use of leftovers. Feeding mainly leftovers may require changes in breeding and feeding strategies, and changes in livestock housing systems. Optimal use of leftover streams enables the livestock sector to produce protein while avoiding competition for land between feed and food production and, therefore, makes an important contribution to future sustainable diets.

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Appendix

A1 Description of vegan diets

The diet composition used as a reference in this study is based on an average of the vegan diet composition of three studies (Table A.1): Van Dooren et al. (2014), Meier and Christen, (2012), and Risku-Norja et al. (2009). These three studies describe western vegan diets that meet common recommendations for a healthy diet. The diet in the Dutch study of Van Dooren et al. (2014) meets the Dutch Dietary Guidelines and is highly comparable to the vegan adjustments of USDA food patterns. In this vegan diet, milk is replaced with soy drinks and extra legumes are included to ensure adequate protein intake. The German study of Meier and Christen, (2012) based diet recommendations on the USDA food patterns because in Germany there were no official guidelines for vegan diets. In this vegan diet, milk is also replaced with soy drinks. We assumed that the soy-based milk contains 12.5% soybeans. The Finnish diet of Risku-Norja et al. (2009) was nutritionally balanced in terms of reasonable daily intakes of carbohydrates, fats, and protein. In the Finnish study an oat-based milk was introduced, corresponding to 100 grams extra oat per person per day.

Table A.1. Average composition of the vegan, based on three papers: Van Dooren et al. (2014), Meier and Christen (2012) and Risk-Norja et al. (2009).

Product group	Van Dooren et al., 2013	Meier and Christen, 2012	Risku-Norja et al., 2009	Average g/d
Vegetables	400	245	268	304
Legumes	21	154	17	64
Fruit	200	250	362	271
Bread	210	-	-	70
Cereal grains	53	295	404	251
Potatoes	105	107	250	154
Nuts and seeds	-	26	-	9
Vegetal oils, margarine	45	27	49	40
Sugar	-	32	60	31
Plant-based drinks	450	732	-	49
Meat replacer	43	-	-	14

A2 Land use of vegan diet

The land use of the vegan diet described by Van Dooren et al. (2014) was 792 m^2 and for the vegan diet described by Meier and Christen (2012) 1052 m^2. No estimate of land use was given for the vegan diet described by Risku-Norja et al. (2009). Nevertheless, the calculations of land use were based on high yields in developed countries. If we account land use of the average vegan diet (see Table A.1) based on global average yields, land use will be 0.13 ha per person. Global average yields are based on Monfreda et al. (2008) and are presented in Table A.2. Adopting this 0.13 ha as average land use of a vegan diet and we know that 0.16 ha of arable land and permanent cropland is available per person, it seems possible to feed the world human population a vegan diet in 2050. Note: the world population is projected to reach 9.7 billion by 2050 (UN, 2015). We, however, want to make some remarks related to the assumption of land use. First, the estimation of the land use does not include food-waste. In case 10% of our food is wasted we need 0.14 ha and 0.16 ha in case of 30% food-waste (see below for information related food-waste). Second, in addition to crop production for human food, arable land and permanent cropland are needed for other functions such as the production of clothes. Third, these vegan diets are formulated based on health recommendations and, therefore, do not represent the total feed intake e.g. do not include (luxury) snacks and drinks. Hence, total feed intake probably results in a higher land use. Fourth, there are large variations in estimating the area available for crop production and pasture. Ramankutty et al. (2008), for example, indicated that there were 1.5 billion ha of cropland (95% confidence range of 1.22-1.71) and 2.8 billion ha of pasture (95% confidence range of 2.36-3.00) worldwide available in 2000. Finally, when leftover streams are used to produce animal source food (ASF), part of the products in the vegan diet e.g. soy milk or legumes can be replaced by the produced ASF resulting in a reduced land use.

Table A.2. Global average yields based on Monfreda et al. (2008).

Product group	Ton/ha/harvest
Cereals	3.1
Oil crops	2.4
Forage	17.6
Pulses	1.1
Roots and tubers	17.7
Fruit	10.5
Vegetables	17.1
Fiber	1.7
Sugar crops	56.8
Three nuts	1.2
Other crops	6.7

A3 Assessing the amount of available co-products and food-waste

To calculate the amount of co-products and food-waste available, we first determined the main product(s) used in each product group, based on Gustavsson et al. (2011). For each main product we determined the production process to determine the co-products related to the production of the main product. We based this on documentation reports of Feedprint (Vellinga et al., 2013). Table A.3 shows the co-products that become available during the production of the average vegan diet. The availability of specific co-products depends on the assumptions made for the main products. For example, we assumed that the main product used in the product group 'vegetal oils and margarine' is soybean oil. However, in Europe sunflower seed and rape seed are the main products while soybean is the main product in North America, Oceania, and industrialised Asia. Our results, related to the amount of pork will change, in case we assume sunflower oil is used, because of the higher fat content of sunflower seeds and the lower nutritional value of sunflower seed meal compared to soybeans and soybean meal.

During the processing and consumption of food, about one third is wasted according to the FAO (Gustavsson et al., 2011). In developed countries people throw away 95-115 kg food per year. Food is spilled mainly when production exceeds consumers demands and during the consumption stage when people throw food away which is still suitable for human consumption. In developing countries 6-11 kg of food is wasted (compared with the 95-115 kg in developed countries). This is mostly due to e.g. technical limitations and limited available

Table A.3. Co-products of the annual production of the average vegan diet for one person as described in Table A.1.

Co-products, fresh basis	kg/person/year
Molasses	2
Potato cuttings	2
Potato peels	1
Potato starch, dried	1
Soybean hulls	9
Soybean meal	55
Sugar beet pulp	15
Wheat bran	19
Wheat germ	3
Wheat middlings	20
Total	129

Table A.4. The amount of food-waste available for animal production, based on the assumption that 10% of the average vegan diet is (Table A.1.) wasted.

Food-waste	kg/person/year
Apples	10
Bread meal	3
Potato chips	3
Potatoes	3
Soybeans	6
Sugar	1
Soy oil	2
Vegetables	11
Wheat flour	10
Total	46

infrastructure. To reduce the environmental impact it is essential to reduce the amount of food-wasted as all food-waste results in a loss of resources and unnecessary environmental impact. Nevertheless, it is unlikely to entirely prevent waste of food and, therefore, we assumed that a part of our food will always be wasted. These products can be used as livestock feed. In order to estimate the amount of food-waste available for livestock we assumed that 10% of the vegan diet (Table A.1) is wasted (Table A.4).

We assume that the co-products and food-waste of the vegan diet are fed to pigs (see main paper for explanation). However, some co-products and waste products are less suitable for pigs, because of their low digestibility in monogastrics or because of limitations of the feeding system. We, therefore, did not take those products (vegetables, raw potatoes, and fruit) into account.

A4 Assessing the nutrient content of co-products and waste products used as pig feed

To estimate the nutrient content (Table A.5.) of one kg of feed based on co-products and food-waste, a commercial linear programming tool (i.e. Bestmix®, Adifo, Maldegem, Belgium) with CVB (2010) database of feed ingredients was used. The diet composition was for almost 99% based on the use of the co-products and waste products in the available ratio (Table A.3. and Table A.4.). One percent was left to add a premix to provide minerals and vitamins, including limestone, and salt.

Table A.5. Diet and nutritional composition of pig feed, based on the use of co-products (Table A.3) and waste products (Table A.4.) in the available ratio on product basis (with the exception of wet products, which were recalculated to a DM content of 880 g/kg).

Ingredients	%	Nutrient content	g/kg
Soybean meal RC<45 RC<480	37.0	Dry matter content	880
Wheat middlings	13.9	Net energy, MJ	8.27
Wheat bran	13.4	Lysine (SID[2])	12.7
Wheat feed flour	7.0	Methionine (SID)	3.3
Soybean hulls RC 320-360	6.5	Cysteine (SID)	3.4
Sugar beetpulp <100	5.3	Threonine (SID)	8.0
Soybeans heat treated	3.8	Tryptophan (SID)	2.9
Wheat grem	2.2	Phosphorus	6.2
Potato cut pre fried	2.2	Crude protein	261
Bread meal	2.0	Crude fat	48
Sugar beet molasses	1.7	Crude fibre	76
Salt	1.3		
Oil (soy)	1.1		
Potato starch (dried)	0.8		
Sugar	0.9		
Premix[1]	0.4		
Potato peels steamed	0.5		

[1] Including 500 FTU of microbial phytase to enhance phytate degradation and phosphorus digestibility

[2] SID, standardised ileal digestible

A5 Assessing the amount of protein from pigs fed with co-products and waste products

In order to calculate the amount of protein from pig meat we used the energy and lysine required to produce a growing-pig of 116 kg calculated by Van Zanten et al. (2015b). In addition, feed is needed for piglet production. Piglet production includes rearing gilts, sows, and their piglets needed for the production of growing pigs. The energy and lysine for growing-pigs, piglets, gilts, and sows in the required ratio as based on Van Zanten et al. (2015b) is summarized in Table A.6.

Table A.6. Energy (NE) and digestible lysine required to produce a growing-pig of 116 kg, for the required piglet and the related sows and gilts (Van Zanten et al., 2015b).

	Feed intake	NE (MJ) g/kg	LYS g/kg	NE (MJ)	Lysine, g	Lysine/MJ
Growing-pig	226	9.59	7.59	2 167	1 715	0.79
Piglets	30	9.68	11.70	290	315	1.08
Gilt	6.7	9.24	8.99	62	32	0.60
Sow	40	9.06	7.42	362	297	0.82

The results in Table A.5 and Table A.6 show that energy is the limiting nutrient. In total 2878 MJ NE is needed to produce one growing pig and 1215 MJ NE is available from the composed feed based on co-products and food-waste. So in total 0.42 pig produced, equal to 49 kg live weight of pig per person per year (0.42 × 116 kg slaughter pig). Using a conversion factor of 0.53 from live weight to edible product and 0.19 from edible product to edible protein (de Vries and de Boer 2010), an estimated 14 grams of pork protein is available per person per day.

We acknowledge that the energy concentration of the feed (8.27 MJ NE/kg) is relatively low and may limit the energy intake and growth rate of the growing-pigs (Quiniou and Noblet, 2012). Thus, it may be questioned whether growing-pigs are able to realise the same growth performance with the feed containing co-products and food-waste as growing-pigs fed with conventional feed. Nevertheless, feed intake capacity and optimal energy concentration differ between growing-pigs, piglets, gilts, and sows. Hence, optimizing the diet composition for each of the different groups of pigs and targeted allocation of co-products and food-waste can be used to optimise the conversion of feed to pork.

A6 Assessing the amount of protein from ruminants grazing on marginal land

The model of Herrero et al. (2013) was used to calculate the amount of protein available per tropical livestock units (TLU) from marginal lands. Herrero et al. (2013) assessed the global number of TLU over several regions of the world. For each region of the world it was defined whether ruminants systems were 100% grass-based. Table A.7 shows the % of dairy cattle, beef cattle, and sheep and goat per ha on 100% grass-based systems on marginal land worldwide and the related average protein production from milk and meat per TLU. Based on this, we calculated the average amount of protein from milk and meat per TLU produced on 100% grass-based systems on marginal land. Average protein production per TLU was 14.14 kg per year (the factor 0.19 to convert from kg edible meat product to kg protein was used and 0.03 to convert from kg milk to kg protein for dairy cattle and 0.04 for sheep and goats). Livestock density was calculated with the model of Herrero et al. (2013) and was 0.5 TLU per ha on marginal land. The area of marginal land, based on GAEZ, was 1.6 billion ha. Based on the above mentioned assumptions, 3 gram of protein per person per day can be produced in 2050 (((14.14 kg protein per TLU×0.5 TLU per ha×1.6 billion ha)/9.7 billion people×1000)/365 days).

Grazing occurs also in other areas besides marginal lands. Those areas currently used for grazing are to a certain extend suitable for crop production (Alexandratos and Bruinsma, 2012). Expanding the area for crop production in these areas will, therefore, lead to a reduction of grazing land. Although these areas are to a certain extend suitable for crop production, they are not yet in use for crop production. We, therefore, made a second calculation in which we assumed all 3.34 billion ha of permanent meadows and pasture are used for 100% grass-based systems.

Tale A.7 shows the % of dairy cattle, beef cattle and sheep and goat per ha on 100% grass-based systems on grassland and the related average protein production from milk and meat per TLU. Protein production per TLU was 14.47 kg per year (average of dairy cattle, beef cattle and sheep and goats). Livestock density - 0.5 TLU per ha - was based on the density of TLU on marginal land. Furthermore, Smil (2014) as well assumed that a livestock density of 0.5 TLU per ha is maximal to prevent degraded grasslands due to overgrazing. Based on the above mentioned assumptions, 7 gram of protein per person per day can be produced in 2050 (((14.47 kg protein per TLU×0.5 TLU per ha×3.34 billion ha)/9.7 billion people×1000)/365 days).

Table A.7. Percent (%) of tropical livestock units (TLU) in 100% grass-based ruminant systems with their related protein production.

	TLU %	Milk protein (kg/TLU/year)	Meat protein (kg/TLU/year)	Protein/%TLU (kg/TLU)
Marginal land				
Dairy cattle	20	17.25	6.32	4.68
Beef cattle	60	-	6.8	4.08
Sheep and goat	20	15.53	11.13	5.38
Total grassland				
Dairy cattle	5	37.81	7.39	2.24
Beef cattle	86	-	10.90	9.37
Sheep and goat	9	17.98	13.48	2.86

Chapter 9

Discussion

1 Introduction

Production of food especially of animal-source food (ASF), has re-emerged at the top of the global political agenda, driven by two contemporary challenges: the challenge to produce enough nutritious food to feed a growing and more prosperous human population and the challenge to produce this food in an environmentally sustainable way. Consumption of ASF generally results in a greater environmental impact than consumption of plant-source food (Meier and Christen, 2012; Scarborough et al., 2014; Hallström et al., 2015). Reducing the environmental impact of ASF can be realized by implementing mitigation strategies, that address the production-side or the consumption-side. Production-side strategies focus on reducing the environmental impact per kg of ASF produced, whereas consumption-side strategies focus on changing consumption patterns by reducing or avoiding consumption of ASF, or shifting from ASF with a higher environmental impact (e.g. beef) to ASF with a lower environmental impact (e.g. pork or chicken) (Wirsenius et al., 2010; Nijdam et al., 2012; Hallström et al., 2015). This thesis focussed primarily on production-side strategies.

Most of the environmental impact of livestock production is related to feed production (De Vries and De Boer, 2010). One strategy to reduce the environmental impact related to feed production is the use of products that humans cannot or do not want to eat (Elferink et al., 2008), such as co-products, food-waste, and biomass from marginal lands for livestock feed (referred to as ‘leftover streams’ in this thesis). This is an effective strategy, because feeding leftover streams to livestock transforms an inedible stream into high-quality food products, such as meat, milk, and eggs. Several mitigation strategies related to the use of leftover streams as livestock feed can be explored. In this thesis, I focused on two production-side strategies: replacing soybean meal (SBM), a feed ingredient with a high environmental impact, with rapeseed meal (RSM), a co-product from bio-diesel industry in pig diets (Chapter 3 and 6); and replacing SBM with waste-fed housefly larvae meal in pig diets (Chapter 5 and 6).

To gain insight into the status quo of the environmental impact of each mitigation strategy, I first used the most commonly used life cycle assessment (LCA) method, called attributional LCA (ALCA). It was my objective, however, to assess the environmental impact of the use of leftovers, while also accounting for two factor: product-packages[1] and feed-food competition[2].

[1] Product-packages refer to a multiple-output situation, e.g. during the processing of sugar beet, not only is sugar produced, but beet-pulp and molasses are also produced. Sugar, beet-pulp, and molasses together form a ‘package of products’ that cannot be produced independently from each other.
[2] Feed-food competition refers to the competition for resources, such as land for crop production for human consumption and land for crop production for livestock feed.

An ALCA, however, does not account for these two factor. To account for environmental impact related to product-packages, I developed a consequential LCA (CLCA) framework of using co-products as livestock feed (Chapter 4). This framework was applied first at the level of an individual feed ingredient (Chapter 4 and 5), and second at the level of the entire pig diet (Chapter 6). To account for feed-food competition occurring in current livestock systems, we developed a method called the land use ratio (LUR) (Chapter 7).

The LUR allows for identification of livestock systems that contribute to the global food supply by minimizing feed-food competition. A ratio, such as our LUR or the protein or energy conversion ratio (Wilkinson, 2011), however, does not provide information about the absolute amount of ASF that can be produced by using only leftover streams. The last mitigation strategy, therefore, focused on how much ASF could be consumed by humans, in a situation where livestock was fed only on leftover steams (Chapter 8). This innovation requires a change not only on the production-side, but also on the consumption-side and was referred to, therefore, as a consistency strategy.

Altogether, I was able to answer both objectives of this thesis: I developed theoretical frameworks that enable evaluation of environmental consequences of using leftovers as livestock feed, while accounting for product-packages and feed-food competition: and I assessed the environmental impact of innovations related to using leftovers, while applying these newly-developed theoretical frameworks.

In the next sections, results will be discussed in relation to the two objectives: first, results related to production-side strategies (section 2); second, results related to the LUR (section 3); third, results related to the consistency strategy (section 4); fourth, the potential of implementing the three innovations (section 5); and finally, conclusions (section 6).

2 Production-side strategies

In this section, the two production-side strategies (replacing SBM with RSM or with waste-fed larvae meal) were assessed, by using either an ALCA or the new framework for CLCA (Figure 1).

2.1 Attributional LCA results

Replacing SBM with RSM

Replacing SBM with RSM in finishing-pig diets was assessed, because RSM became increasingly available following an increase in bio-energy production in the EU. In this strategy, therefore, the RSM content in livestock diets increased at the expense of SBM. Results of an ALCA along the entire production chain (Chapter 3) showed that replacing SBM with RSM in finishing-pig diets hardly changed global warming potential (GWP) (<1%) and energy use (EU) (between 0%-2%), but decreased land use (LU) up to 16%. Feed production had the largest environmental impact: 50%-52% for GWP, 60%-61% for EU, and 77%-80% for LU per kg body weight. These results explain why I focused mainly on the impact related to feed production per kg of body weight gain in follow-up chapters. The ALCA results in Chapter 6, therefore, are slightly different but still resulted in the same conclusion: GWP still hardly changed (between -2%-3%), EU still hardly changed (-1%-1%), but LU decreased (up to 16%).

It is worth noting that results of the sensitivity analysis of the pig production chain in Chapter 2 showed that changes in feed conversion ratio (FCR) and emissions from manure management affected the results most. In Chapter 3, therefore, we also performed sensitivity analysis related FCR and manure management of the pig production chain. Results showed that uncertainty of the FCR and manure management did not have an effect on the relative difference between the scenarios and, therefore, I did not perform a sensitivity analysis related to FCR and manure management in Chapter 6. I realize, however, that dietary changes might affect feed intake, resulting in different FCR, and that changes in the amount of N or non-digestible polysaccharides might affect emissions from manure management (Aarnink, 2012; Quiniou and Noblet, 2012; Mosquera et al., 2013; Hou et al., 2015; Hou et al., 2016).

Replacing SBM with waste-fed larvae meal

Replacing SBM with waste-fed larvae meal was assessed because recent developments indicated environmental benefits of rearing insects as livestock feed, which suggest that waste-fed insects might become an important alternative feed source in the future. No environmental assessments, however, were performed regarding use of waste-fed larvae meal. We, therefore, first had to assess the environmental impact of producing larvae of the common housefly which were grown on poultry manure and on food-waste. Comparing waste-fed larvae meal with SBM, each containing a high protein content, on the basis of feed ingredient level resulted in a greater GWP (lower in case land use change emissions were included) and EU, but less LU per ton of waste-fed larvae meal. Energy use, mainly for on farm heating, was the main contributor to the environmental impact of larvae meal production.

Although waste-fed larvae meal and SBM are both protein rich feed ingredients their nutritional value differs. Based on the results per kg of waste-fed larvae meal, we were able to extend the environmental assessment to the level of the pig diet. Results showed that EU hardly changed (1%), but GWP (29%) and LU (54%) decreased per kg of pig body weight gain, when you replace SBM with larvae meal in pig diets. The assessment at the level of pig diet compared with level of a feed ingredient showed that waste-fed larvae meal was an ingredient with a high nutritional quality (not only of protein, but also of crude fat) and, therefore, enabled the inclusion of more co-products in the diet, resulting in a reduced environmental impact.

To sum up, each mitigation strategy, resulted in a similar or reduced environmental impact. Replacing SBM with waste-fed larvae meal, however, was more promising than replacing SBM with RSM.

2.2 Consequential LCA results

Replacing SBM with RSM

The CLCA results for replacing SBM with RSM showed mainly an increase in GWP (-3-15%), increased EU (1%-12%), and LU (8%-10%) per kg body weight gain[3]. Differences in results between ALCA and CLCA can be explained mainly by differences in the method of assessing the environmental impact of co-products. In RSM S4, for example, 15% SBM and 8% barley were replaced with 23% RSM, and 2% animal fat. Using RSM in pig diets resulted in overall increased use of SBM (the decrease in SBM in pig diets was less than the increase of SBM needed to replace RSM in diets of dairy cows), and using animal fat in pig diets resulted in an overall increased use of palm oil (the decrease in palm oil in pig diets was less than the increase in palm oil needed to replace animal fat in broiler diets). The net environmental impact of this replacement is, therefore, negative because the environmental benefit was less than the environmental costs of using co-products.

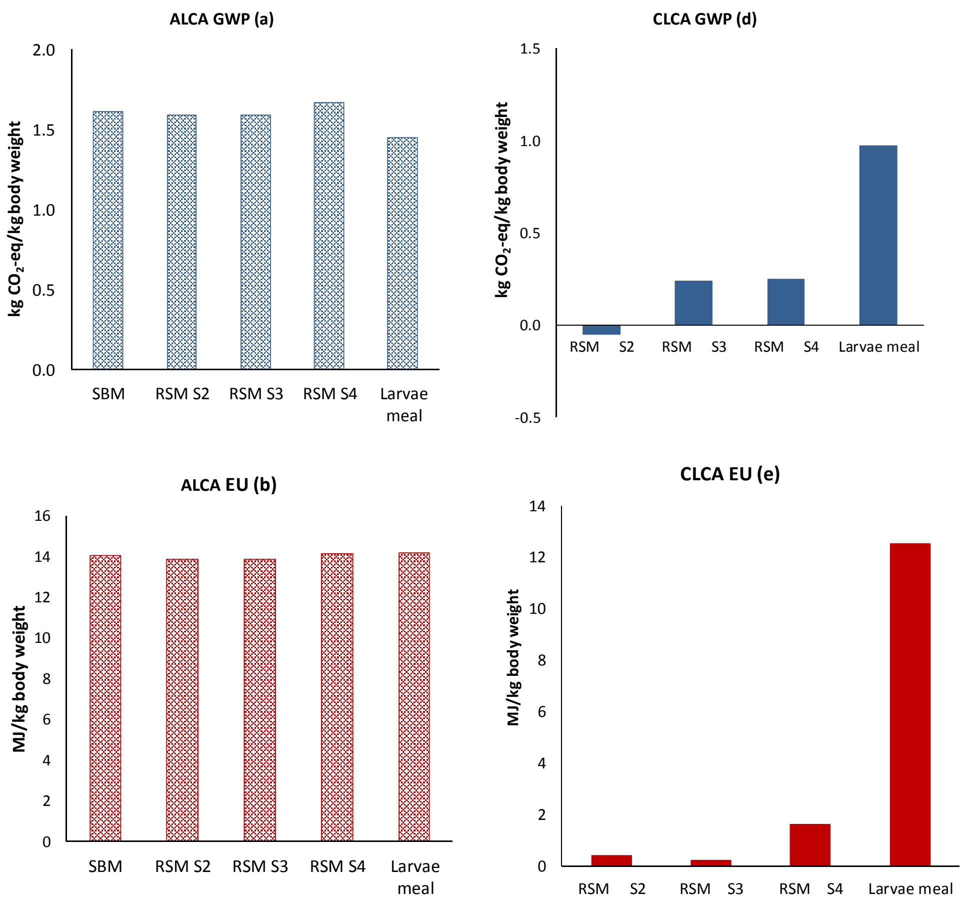
[3] In Chapter 3, three RSM scenarios were assessed. Compared with the SBM diet: the RSM S2 diet had a reduced nutritional value per kg feed resulting in an increased feed intake; the RSM S3 diet had a reduced nutritional value per kg feed resulting in decreased growth performance; and the RSM S4 diet had a similar nutritional value per kg feed. In Chapter 6 only RSM S4 was used. To compare results between chapters, other scenarios related to RSM S2 and S3 were calculated also, based on the same methodology and data as explained and used in Chapter 3 and 6. The only additional data needed was feed intake for S2, 193 kg during the finishing phase, and body weight gain for S3, 68.35 kg during the finishing phase.

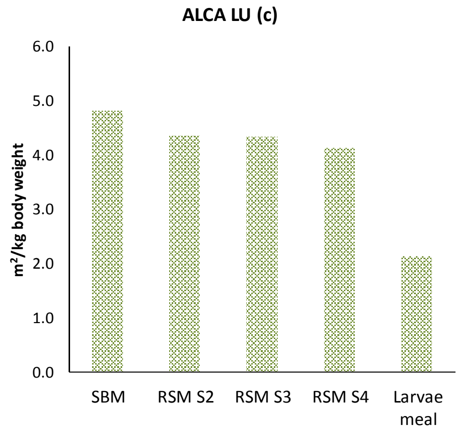
Replacing SBM with waste-fed larvae meal

The CLCA results for replacing SBM with waste-fed larvae meal showed increased GWP (60%) and EU (90%), but LU (73%) decreased per kg body weight gain. The difference in results between ALCA and CLCA was caused mainly by the difference in impact of the waste-fed larvae meal. Food-waste to feed larvae was used initially to produce bio-energy via anaerobic digestion. In a CLCA, the environmental impact related to replace the bio-energy function of food-waste was included (i.e. indirect environmental impact). The net environmental impact became negative, because environmental benefits of replacing SBM with waste-fed larvae meal were less than environmental costs related to the marginal energy source, i.e. fossil-energy, replacing the bio-energy, which was initially produced with the food-waste. Results of the indirect environmental impact, however, are situation specific: if the marginal energy source were wind or solar energy, the net environmental impact of using larvae meal could be positive. Waste-fed larvae meal, therefore, appears to be an interesting mitigation strategy only when energy from wind and solar energy are used more dominantly than energy from fossil sources. Results showed that assumptions required to perform a CLCA, such as defining the marginal product[4], therefore, are space and time specific and can have a large impact on the final results. Results of a CLCA, therefore, seem to be not only relatively more uncertain than results of the ALCA but also more exact.

To sum up, each mitigation strategy, resulted in an increased environmental impact, except for LU for waste-fed larvae meal.

[4] The marginal product is the product that responds to a change in demand. An increased demand for a co-product (e.g. RSM) results in an increased production of the marginal product (e.g. SBM).


ALCA GWP (a)
kg CO_2-eq/kg body weight
2.0
1.5
1.0
0.5
0.0
SBM
RSM S2
RSM S3
RSM S4
Larvae meal
CLCA GWP (d)
kg CO_2-eq/kg body weight
1.5
1.0
0.5
0.0
-0.5
RSM S2
RSM S3
RSM S4
Larvae meal
ALCA EU (b)
MJ/kg body weight
16
14
12
10
8
6
4
2
0
SBM
RSM S2
RSM S3
RSM S4
Larvae meal
CLCA EU (e)
MJ/kg body weight
14
12
10
8
6
4
2
0
RSM S2
RSM S3
RSM S4
Larvae meal



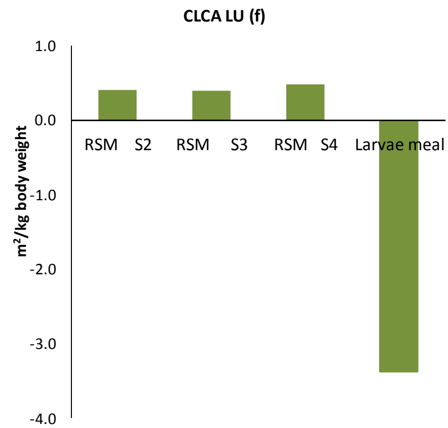


Figure 1. Global warming potential (GWP), energy use (EU), and land use (LU) for the mitigation strategies replacing soybean meal (SBM) with rapeseed meal (RSM) or with waste-fed larvae meal, calculated with the attributional life cycle assessment (ALCA) (Figures a, c, and e) or with the consequential life cycle assessment (CLCA) (Figures b, d, and f). The RSM strategy contained three scenarios. Compared with the SBM diet: the RSM S2 diet had a reduced nutritional value per kg feed resulting in an increased feed intake; the RSM S3 diet had a reduced nutritional value per kg feed resulting in decreased growth performance; and the RSM S4 diet had a similar nutritional value per kg feed.

2.3 Value of ALCA versus CLCA

The difference in ALCA and CLCA results was because CLCA considers the environmental consequences of using product-packages, whereas ALCA does not. Looking at the environmental impact of pig production based on an ALCA, one would advise feed companies, for example, to increase use of co-products and use of waste-fed larvae meal. This advice is because co-products and food-waste have a relatively low economic allocation, therefore, a relatively low environmental impact. Increasing use of co-products, for example, therefore, leads to a reduced environmental impact, based on an ALCA.

Results of the CLCA study, however, showed that increasing the use of co-products or food-waste, which already have an application in a different production system, in animal feed does not necessarily result in a total reduction of the environmental impact. That the environmental impact is not decreasing is because co-products and food-waste are available only in limited supply. Increasing their use in animal feed, consequently results in a reduced use of co-product or food-waste in another sector, requiring replacing them with a different marginal product. The environmental impact of increasing the use of a co-product or food-waste, therefore, depends on the net environmental impact. The net environmental impact is the environmental benefits minus the environmental costs. Environmental benefits on the one hand are determined by the decrease in environmental impact related to the product that was replaced with co-products or food-waste. The environmental costs on the other hand are determined by the increased environmental impact related to the marginal product, i.e. product that replaces the 'old' application of the co-product or food-waste.

For each mitigation strategy assessed in this thesis, the environmental impact did not change or increased when CLCA was applied. This results would have been overlooked if results had been based solely on ALCA.

Conclusion. The CLCA framework developed in this thesis allows handling of product-packages. Because this method includes the environmental consequences related to product-packaging, I found contradictory results when I compared the CLCA method with the standard ALCA method. Based on the ALCA method, each mitigation strategy was promising at the level of the pig diet (waste-fed larvae more so than RSM). Based on the CLCA method, however, each strategy did not change the environmental impact and, in some cases, even resulted in an increased environmental impact. This result shows that consideration of the environmental consequences of related product-packaging is essential.

To gain insight into the environmental impact of feed, animal nutritionists can use an ALCA. If policy makers or the feed industry, however, want to assess the net environmental impact of a potential mitigation strategy, it is recommended to perform a CLCA. The framework developed in this thesis can be used to perform such an assessment.

3 Land use ratio

A land use efficiency method called land use ratio (LUR) was developed to deal with feed-food competition (Chapter 7). LUR is defined as the maximum amount of human digestible protein (HDP). That is derived from food-crops on all land used to cultivate feed required to produce one kilogram ASF over the amount of human digestible protein (HDP) in that one kilogram ASF. An LUR <1.0 implies that livestock produce more HDP per m^2 than crops produce. Only if LUR is zero, which can occur by feeding only co-products and food-waste, for example, or by cows grazing only on marginal land, is feed-food competition completely avoided. In terms of global food supply, however, livestock systems with a LUR <1 are more efficient than crops because they produce more HDP per m^2 than crops do produce.

In Chapter 7, the LUR was applied to three case studies: dairy cows on peat soil, dairy cows on sandy soil, and laying hens. These three systems were chosen because they have comparable results related to land use, when calculated with an ALCA. The LUR's for the case of laying hens and dairy cows on sandy soil equalled about 2.1, implying that all land required to produce 1 kg HDP from laying hens or cows on sandy soil could yield about twice the amount of HDP from human food-crops. For dairy cows on peat the LUR was 0.67. The LUR for cows on peat was lower than for cows on sandy soil because land used to grow grass and grass silage for cows on peat was unsuitable for production of food-crops. Results of the dairy cows on peat soil, demonstrate that livestock can produce HDP more efficiently than crops do.

For this discussion, LUR of the two mitigation strategies related to pig production were also calculated (see Figure 2). Looking at each mitigation strategy, the RSM strategy (3.9-4.0), and the waste-fed larvae meal strategy (2.7) had a lower LUR than the basic scenarios containing SBM (4.6). Pigs, however, had a greater LUR than dairy cows (on peat and sandy soils) and laying hens.

The LUR results demonstrated clearly that livestock systems, e.g. dairy cows, that value land less suitable for arable production (marginal land) can produce HDP more efficiently than crops can. The LUR results related to the pig production systems, which use co-product (RSM) and food-waste (waste-fed larvae meal), however, did not produce HDP more efficiently than crops did. This result is mainly because the amount of co-products was low, in the pig diets we considered; the SBM scenario contained 97% determining products , the RSM scenarios 72%, and the waste-fed larvae meal scenario 56%. By increasing the use of co-products in diets of pigs one might end up with a LUR <1.0.

It was difficult, however, to define the amount of land used for the production of the co-products. The LUR used economic allocation to determine LU. The LU related to sugar beets, for example, was allocated to the various products, e.g. sugar, molasses, and beet pulp, based on their relative economic values. The greater the relative economic value of the co-product, the greater the portion of LU related to the cultivation of the main product, e.g. sugar beets.

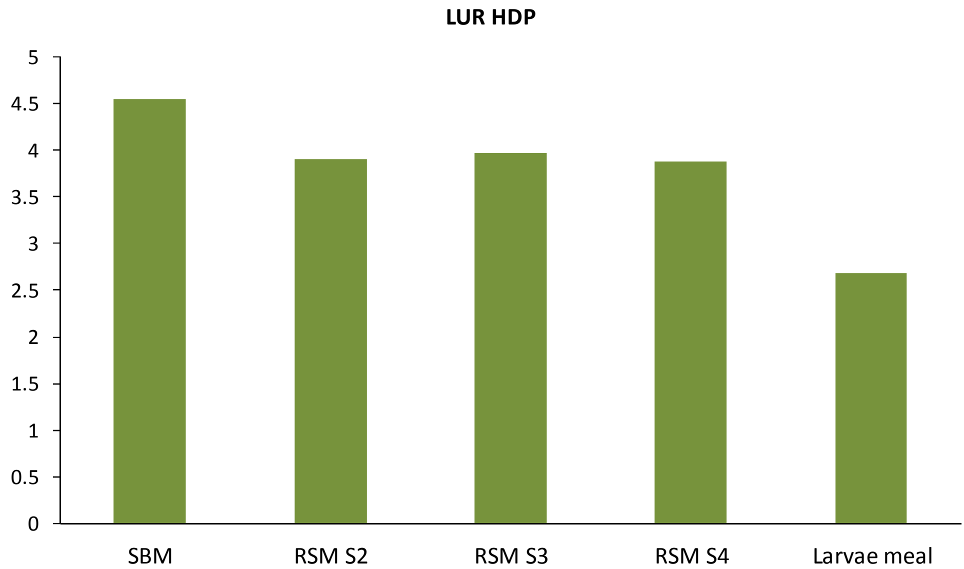


Figure 2. Land use ratio (LUR) in kg human digestible protein in crops/ human-digestible protein (HDP) in animal source food, applied for the mitigation strategies replacing soybean meal (SBM) with rapeseed meal (RSM) or with waste-fed larvae meal in finishing-pig diets. The RSM strategy contained three scenarios. Compared with the SBM diet: the RSM S2 diet had a reduced nutritional value per kg feed resulting in an increased feed intake; the RSM S3 diet had a reduced nutritional value per kg feed resulting in decreased growth performance; and the RSM S4 diet had a similar nutritional value per kg feed.

The current market value of co-products or food-waste, however, reflects the value of these products for animal feed. The choice for economic allocation to determine the LU of co-products, therefore, is not consistent with the way marginal land is treated in the LUR, because this was based on its suitability for food-crop production.

I would have preferred an allocation method in which the market value of co-products and food-waste reflected their nutritional value for humans. This would have meant that co-products , for example, that are not consumed by humans (i.e. products that are not suitable or not wanted by humans) would have a zero allocation and, therefore, no land use. In the current system, however, feed and food markets are interconnected. It is not possible, therefore, to make a distinction between feed and food based on their nutritional value for humans.

Although, the choice for economic allocation was not optimal, it appeared to be the best available alternative. Besides economic allocation, allocation can be based, for example, on

HDP or energy (HDE). Allocation based only on HDP, however, resulted in the problem that crops yielding energy- and protein- products (e.g. soybean oil and SBM) produce less protein compared with crops yielding only protein products (e.g. peas). Such a problem might be solved by using a nutrient density score. The nutrient density score of a food product relates the nutrient content of 100 g or 100 kcal of a product to the daily recommended intake, and averages the values of different nutrients into one final score (Drewnowski and Fulgoni, 2014). Despite a low nutrient density score, however, an individual food item can be valuable at the dietary level because of its richness in one very scarce nutrient.

An alternative to using economic allocation is to assume that co-products do not have nutritional value for humans. Such an assumption implies that land use of crop production is allocated only to those determining products, and co-products that do not require any land. To show the impact of such an assumption on the results, I made a calculation for the discussion based on zero allocation. The results of that calculation is that the LUR for the SBM scenario decreased from 4.6 to 4.4, RSM scenarios from 4.0-3.8 to 3.2-3.1, and waste-fed larvae meal scenario from 2.7 to 2.2. No LUR was <1, because the main part of the pig diets still contained determining products instead of co-products.

Conclusion. For current livestock systems in the Netherlands, my results showed that LUR was highest for pigs, then laying hens, and lowest for dairy cows. Only dairy cows on peat soil that valued land that was not suitable for arable production produced HDP more efficiently than crop production systems did. For the other systems, it was more efficient to produce crops for human consumption instead of producing feed for livestock. The LUR, therefore, helps with the identification of livestock production systems that produce HDP more efficiently than crop production systems do, with an LUR <1. These livestock systems have an important role to play in future sustainable nutrition supply.

4 Consistency strategy

Results of the LUR showed that livestock production systems using mainly co-products, food-waste, and biomass from marginal land, can produce HDP more efficiently than crop production systems do. The availability of those leftover streams is, however, limited and, therefore, the amount of ASF produced based only on leftover streams is also limited. Because LUR is a ratio, LUR results do not give an indication of how much ASF can be produced based on livestock systems that feed mainly on leftover streams. We, therefore, assessed the amount of ASF produced from livestock fed with only leftover streams, in Chapter 8. The calculation was based on the assumption that a vegan diet was consumed in principle, resulting in co-products and food-waste that were fed to pigs and, furthermore, that all biomass produced from grazing land was fed to ruminants. Results showed that in total 21 g ASF protein per person per day could be produced by feeding livestock entirely on leftovers. Of this 21 g, about 17 g was produced without competition between feed and food-crops for arable land (4 g less if grassland with potential for crop production was excluded). Although the calculations in Chapter 8 contained many assumptions, results showed that livestock has an important role in future sustainable diets, in that livestock makes use of products inaccessible to humans.

Main assumptions

Some of the main assumptions made in the calculations were that co-products were not used for human consumption, that only 10% of food will be wasted, and that livestock was allowed to be feed food-waste. That co-products are not used for human consumption was based on the assumption that those co-products and food-waste do not have a value as human food and, therefore, do not have a nutritional value for humans. For example, SBM was assumed to be a co-product produced during the production of soybean oil, which was needed in the vegan diet. The SBM was assumed not to have an economic value as human food. This assumption does not mean, however, that co-products and food-waste cannot make a nutritional contribution to the human diet, but that humans value co-products so low (e.g. due to taste preference) that their value as food is negligible. Whether or not these assumptions will hold true for co-products and food-waste, currently and in the future, is hard to say. Technological developments, for example, might make it possible to up-grade some leftover streams from livestock feed to human food. It might also be possible to up-grade unused leftover streams to livestock feed, e.g. improving the feeding value of straw by using fungi (Khan et al., 2015). Human preferences might also vary with time. The amount of co-products and food-waste available as livestock feed, therefore, might change over time, depending not only on technical developments but also on human preferences.

Another important assumption made in the calculations in Chapter 8 was that co-products and food-waste were fed to pigs, whereas biomass from marginal land was fed to ruminants. Co-products and food-waste also can be used to feed broilers and laying hens. In addition to conventional livestock species, insects also can be used. Insects fed on manure, for example, can be used as livestock feed, and insects fed on food-waste can be used for direct human consumption. Furthermore, my calculations did not include potential protein production, for example, from captured wild fish or from micro- and macro-algae. Total protein production per person per day, therefore, might be even larger than the 21 g that we calculated.

Comparing the results with other consistency strategy studies

Several recent studies have concluded that using leftover streams is important to reduce the environmental impact of ASF (e.g. Schader et al., 2015; Röös et al., 2016), but only four had a consistency strategy approach (Elferink et al., 2007; Smil, 2014; Schader et al., 2015; Van Kernebeek et al., 2015) (table 1). In consistency strategy studies, arable land is not used or uses only minimally to produce feed (LUR<1.0), only products that humans cannot or do not want to eat are fed to livestock, and biomass from marginal land is used to feed ruminants (Garnett, 2009; Röös et al., 2016).

Elferink et al. (2008) concluded that about 27 g protein originating from pig meat can currently be consumed per person per day. Their calculation considered only available co-products, and did not consider food-waste and biomass from marginal land. Availability of co-products was based on average Dutch consumption of three crops: sugar beets, soybeans, and potatoes, which represent approximately 60 % of the co-products produced from the food industry in the Netherlands.. They then calculated that Dutch person consumes on average 43 kg sugar, 18 kg soy oil, and 97 kg potatoes per year. Furthermore, they corrected for the total share of co-products produced in the Netherlands. My calculation in Chapter 8, however, is not based on the current food intake of humans, but on what we should consume from a health perspective. In my calculation, therefore, consumption of sugar (11 kg), soy oil (15 kg), and potatoes (56 kg) were lower compared to Elferink et al. (2008), resulting in lower availability of co-products per person.

Smil (2014) concluded that in total about 200 million tons of meat (carcass weight) can be produced currently, resulting in about 9 g of protein per person per day. He based his calculation on the amount of available co-products, crop-residues, and biomass from grazing land, but he did not include food-waste. He assumed that globally 40 Mt meat can be produced from ruminants feeding on crop-residues, 40 Mt pig meat and 70 Mt chicken meat can be produced from monogastrics feeding on co-products, and 40 Mt meat can be produced from ruminants grazing on grasslands.

Schader et al. (2015) concluded that in 2050 about 26 g of meat, 2 g eggs, and 138 g milk can be consumed per person per day, resulting in protein supply of 9 g per person per day. Their calculation was based on the amount of available co-products and biomass from grazing land, but did not include food-waste. Bottom-up mass flows were used for the calculations, based on data from the Food and Agricultural Organisation.

Van Kernebeek et al. (2015) concluded that land use was most efficient if people (up-to a human population of 35 mln) would consume about 7 g of protein from ASF (mainly milk) derived from livestock fed mainly on co-products. Their calculation was mainly based on co-products and marginal land, and hardly on food-waste. They used linear programming to determine minimum land use required to feed the Dutch population.

Table 1. Estimates of protein production from animal source food from livestock production systems that only use feed products that are not in competition with humans: co-products, food-waste, and biomass from marginal land and crop-residues.

	g protein per capita per day	Food-waste	Co-products	Biomass marginal land	Crop-residues	ASF products
Elferink et al. (2007)[a]	27		x			meat
Smil (2014)	9		x	x	x	meat
Schader et al. (2015)	9		x	x		meat, milk, egg
Van Kernebeek et al. (2015)[a]	7		x			meat, milk
Van Zanten et al. (2015)	21	x	x	x		meat, milk

[a] Based on the Dutch situation, other studies are global

The amount of protein from ASF per person per day calculated by Smill (2014), Schader et al. (2015), and Van Kernebeek et al. (2015) was lower than our calculation. The amount of ASF produced in our calculation was higher because we included not only food-waste, but also feed-food crops. The importance of food-waste as livestock feed was also recognised by Zu Ermgassen et al. (2016), who concluded that feeding heat-treated food-waste to livestock can reduce the land use impact of pork production within the EU by 20% (about 1.8 billion hectares of agricultural land). Feeding food-waste to livestock is currently not allowed, and some people question the legal status of food-waste (Zu Ermgassen et al., 2016). Zu Ermgassen et al. (2016) state that feeding food-waste to livestock can be a safe alternative if food-waste is heat-treated. Such practices are applied commonly in Japan and South Korea, where about 35% of the food-waste is fed to livestock (Zu Ergassen et al., 2016).

Furthermore, we included feed-food crops by choosing those food ingredients in the vegan diet whose co-products had a high nutritional value for livestock. Oil production originated from soy cultivation, for example, resulted in the co-product SBM. Compared with other co-products from oil processing, e.g. sunflower meal, SBM has a high nutritional value for livestock. Elferink et al. (2008) also included SBM as a co-product in their calculation and concluded that about 27 g of protein per person per day could be consumed. This conclusion not only has an impact on the final protein production from pork, but also demonstrates the importance of optimizing crop production based on called feed-food crops.

My results showed, therefore, that including food-waste and considering feed-food crops are important because considering this, increases the amount of ASF that can be produced without feed-food competition. Including food-waste and considering feed-food crops is especially interesting because these strategies are not currently applied. Applying such strategies, therefore, might reduce the environmental impact.

Although using different assumptions, each studies concluded that consuming a small or moderate amount of ASF by humans reduces land use most. At present, the average global consumption of animal protein, however, is about 32 g per person per day. To avoid feed-food competition completely, the total world-wide consumption of ASF must, therefore, be reduced. I did not intend to calculate the amount of ASF people should eat, nor did I intend to calculate the upper human population limit that can be fed on the current amount of arable land available. My calculations, however, show that land use can be reduced by using food-waste and considering feed-food crops in current production systems, and that livestock is important in sustainable nutrition supply.

All studies, except for Schader et al. (2015), focussed only on land use. In this thesis, the main focus was on LU, followed by GWP, and EU. I am aware, however, that besides those environmental impact categories, other environmental impact categories are also important and can result in trade-offs. Schader et al. (2015), however, concluded that feeding only co-products and biomass from marginal land to livestock also resulted in a decrease of GHG emissions, EU, N-surplus, P-surplus, pesticide use, water use, and soil erosion potential.

Production-side versus consumption-side versus consistency strategies

The conclusion that consuming a small amount of ASF is most efficient from a perspective of land use contradicts the conclusions from consumption-side studies (generally based on GHGs and LU) that a vegan or vegetarian diet is most environmental friendly (Meier and Christen, 2012; Scarborough et al., 2014; Hallström, 2015). Consumption-side studies often suggest, furthermore, that shifting the type of ASF from ruminant meat to monogastric meat will reduce the environmental impact (Nijdam et al., 2012). The results of this thesis (Chapters 7 and 8), however, indicate that ruminants can play an important role in converting biomass from marginal land into high-quality protein products.

The contradiction in results between the consistency strategy and the consumption-side strategy can be explained based on Figure 3, which illustrates that the consumption of ASF can decrease or increase land use (Peters et al., 2007), depending on the percentage of ASF consumed in the diet. Consuming a small amount of ASF is most land efficient (green line). No consumption of ASF (decreasing red line in Figure 3) results in a higher environmental impact of land use compared with a small amount of ASF (Van Kernebeek et al., 2015). Fairlie (2010) refers to “default livestock” where the main function of livestock is to use leftover streams optimally. In default livestock systems, feed-food competition is minimized and HDP production is optimized (LUR<1). As soon as the average global consumption of

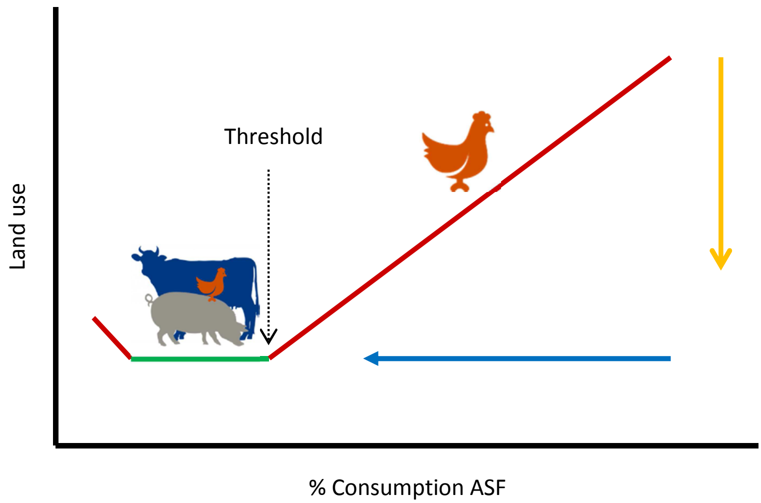


Figure 3. Land use is most efficient when animal source food (ASF) is consumed between 7 g and 27 g of protein per person per day from default livestock (green line). Land use is less efficient when no ASF is consumed (<7 g protein, red line) or after the threshold point (>27 g protein, red line). Land use can be reduced by production-side strategies, aiming to reduce land use per kg of product (yellow arrow) or by consumption-side strategies, aiming to reduce land use by changing consumption patterns (blue arrow).

ASF exceeds the threshold line, feed is in competition with food for arable land (increasing red line). After the threshold point (>27 g protein) the environmental impact of livestock production can be reduced by implementing production-side strategies, which means decreasing the impact per kg product, e.g. by sustainable intensification (yellow arrow). Such mitigation strategies should be applied only from the threshold point onwards. Before the threshold point (>7 and <27 g protein) all livestock species mainly can be fed leftover streams. After the threshold point (>27 g protein), when all leftover streams are used, livestock species can be fed only with products that can also be used directly for human consumption. In current livestock systems, however, feed sources that are from leftover streams and feed sources that are in competition with food-crops are interconnected.

Livestock systems in which feed-food competition is occurring should aim to reduce the impact per kg of ASF. Most production-side studies, therefore, conclude that intensification results in a reduced environmental impact. Havlík et al., 2014, for example, found that intensifying global ruminant systems, by a transition from grazing systems to mixed systems, reduces greenhouse gas emission by 9%, based on expected population growth for 2030 (Havlík et al., 2014).

In addition to implementing production-side strategies the environmental impact can be reduced also by implementing consumption-side strategies, e.g. reducing the consumption of ASF (blue arrow). Research suggested shifting to low environmental impact ASF products, such as chicken meat (Hallström et al., 2015). My results showed that up to the threshold point, ASF can be consumed from monogastrics and ruminants. Several studies have shown that after the threshold point (>27 g protein), chicken meat, eggs, and maybe milk (also implying some beef) have the lowest environmental impact per kg of ASF protein (De Vries and De Boer, 2010).

Conclusion. Current livestock systems can reduce land use by accounting for the use of food-waste as livestock feed and by considering feed-food crops. Livestock fed with co-products, food-waste, and biomass from marginal lands can produce about 21 g of protein per person per day. Livestock, therefore, has an important contribution to sustainable nutrition supply. When all ASF products from default livestock are consumed, production of livestock will result in competition for arable land between feed and food.

5 Implementing potential mitigation strategies

The mitigation strategies replacing SBM with RSM or with waste-fed larvae meal, are both so-called production-side strategies and can be implemented without changing current consumption patterns. Compared with the basic situation, in which SBM was used as a common protein source in pig feed, the incremental innovation of replacing SBM with RSM hardly changed the environmental impact. This incremental innovation is mainly a technical innovation and will require, as do most incremental innovations, little change to be accepted by society (Elzen and Wieczorek, 2005).

The system innovation of replacing SBM with waste-fed larvae meal showed potential to reduce LU. The production of waste-fed larvae meal to feed livestock, however, requires a system innovation. First, it requires technical innovations to build large, automated industrial facilities to raise the larvae that are economically sustainable; second, it requires social acceptance to allow livestock to be fed on insects; and last, it requires institutional changes (when food safety issues are solved) because feeding waste-fed insects to livestock is currently forbidden in the EU (Van Huis, 2015). Since 2013, however, feeding insects to fish in aquaculture is already permitted through EU regulation and using insects as feed for pigs and poultry is currently under consideration. Although the use of insects is still in a developing phase, a new agricultural sector is emerging (Van Huis, 2015).

The results of this thesis showed that changing the application of leftovers did not necessarily result in improved net environmental impact. Whether or not changing the application results in an improved net environmental impact depends mainly on whether or not the environmental “benefits” of using the co-products in its ‘new’ application are higher than the environmental “costs” related to replacing the co-product in its ‘old’ application. Producing waste-fed insects, for example, appeared not to be beneficial because the “cost” of replacing bio-energy (produced with the food-waste) with fossil fuels was higher than the “benefits” of replacing SBM and fishmeal with waste-fed larvae meal.

The consistency strategy of feeding livestock with only leftover streams requires a transition towards a livestock sector that contributes to sustainable nutrition supply. The term “transition” highlights a difference between an earlier and a later stage of livestock production, e.g. horse-power based versus tractor-power based (Elzen et al., 2012). To feed livestock only on co-products, food-waste, and biomass from marginal land, future innovations should focus on adapting livestock systems. Innovations are needed to overcome food-safety problems and technical concerns related to collecting the leftover streams. Feeding mainly leftovers, furthermore, might require changes in e.g. breeding and feeding strategies. Additional to changes in livestock production systems, future innovations should focus as well on changing human diets. By feeding only leftovers to livestock, about 21 g of protein can be produced. Although this fulfils a third of the recommended protein intake, it does not fulfil the current average global consumption of animal source protein of 32 g per person per day (FAO stat, 2015).

Reducing consumption of animal source protein from about 32 g per person per day to about 21 g, might not be a realistic goal for the coming decade (Foley et al., 2011). In the short term if, no major changes in consumption of ASF occur, then production-side strategies and consumption-side strategies are needed to reduce the environmental impact. In the long term, however, mitigation strategies to reduce the environmental impact should focus on consistency strategies. Such a change towards consistency strategies, requires a paradigm shift; livestock production should not focus on increasing efficiency of the animals but on increasing efficiency of the entire food system. Future research on the role of livestock in sustainable nutrition supply should, therefore, assess the entire food production system in relation to consumption of ASF.

6 Conclusion

Livestock production has a major impact on the environment, and feed production is responsible for the majority of this impact. To reduce the environmental impact of livestock production systems, this thesis focused on using products for livestock feed that humans cannot or do not want to eat, such as co-products, food-waste, and biomass from marginal lands. This is an effective strategy, because it transforms an inedible stream into animal source food (ASF) while minimizing feed-food competition for arable land. Three mitigation strategies were applied; two production-side strategies that aimed to reduce the impact per kg of product; and one consistency mitigation strategy that aimed to reduce the impact by combining production-side and consumption-side strategies.

We applied two production-side mitigation strategies: replacing soybean meal (SBM) with rapeseed meal (RSM), and replacing SBM with waste-fed housefly larvae meal in pig diets. Based on the commonly used attributional life cycle assessment (ALCA) method, results showed that each mitigation strategies was promising (waste-fed larvae more so than RSM). The ALCA method, however, did not account for product-packages and feed-food competition.

A consequential theoretical framework was developed to account for product-packages. Results were contradictory compared with the ALCA method. Based on the consequential LCA (CLCA) method the two strategies hardly change the environmental impact and, in some cases, even resulted in an increased environmental impact. Accounting for product-packages increased the environmental impact of each strategy. Producing waste-fed larvae, for example, appeared to be not beneficial, because the cost of replacing bio-energy produced with food-waste, with fossil fuels was greater than the benefit of replacing SBM with waste-fed larvae meal. If results were based solely on ALCA, then these potentially negative impacts would have been overlooked. Consideration of the environmental consequences of product-packaging, therefore, is essential. If policy makers or the feed industry want to assess the net environmental impact of a potential mitigation strategy, then I recommend to perform a CLCA instead of a ALCA. The framework developed in this thesis can be used to perform such an assessment.

The land use ratio (LUR) was developed to account for feed-food competition. Results of the LUR illustrated that dairy cows on sandy soils, laying hens, and pig production systems in the Netherlands have a LUR >1. This means, in terms of protein produced per m^2, that it is more efficient to produce crops for direct human consumption than produce feed for livestock. Only dairy cows on peat soil produced human digestible protein (HDP) more efficiently than crops did, because peat soil are not suitable for crop production. The LUR allows identification of livestock production systems that are able to produce HDP more efficiently than crops do. Such livestock systems (with a LUR <1.0) have an important role to play in future sustainable nutrition supply.

The LUR, however, does not provide information about the amount of ASF produced from livestock. The consistency mitigation strategy, therefore, focused on the amount of ASF that can be consumed by humans, when livestock are fed on leftover steams (so-called default livestock). Up to 21 g of protein per person per day can be consumed, while avoiding competition between feed and food for arable land. Within current livestock systems, co-products and biomass from marginal land are already used. Feeding food-waste and considering feed-food crops, however, are examples of mitigation strategies that currently can be implemented to reduce further the environmental impact.

On average, it is recommended to consume about 57 g of protein from ASF or plant-origin per person per day. Although ASF from default livestock does not fulfil the current global protein consumption of 32 g per person per day, about one third of the protein each person needs can be produced without competition for land between feed and food production. Livestock, therefore, does have an important contribution to the future nutrition supply.

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Summary

Production of food has re-emerged at the top of the global political agenda, driven by two contemporary challenges: the challenge to produce enough nutritious food to feed a growing and more prosperous human population, and the challenge to produce this food in an environmentally sustainable way. Current levels of production of especially animal-source food (ASF), pose severe pressure on the environment via their emissions to air, water, and soil; and their use of scarce resources, such as land, water, and fossil energy. The livestock sector, for example, is responsible for about 15% of the global anthropogenic emissions of greenhouse gases and uses about 70% of global agricultural land.

Many proposed mitigation strategies to feed the world sustainably, therefore, focus primarily on reducing the environmental impact of the livestock sector, so-called production-side strategies. Other strategies focus on changing consumption patterns by reducing consumption of ASF, or on shifting from ASF with a higher environmental impact (e.g. beef) to ASF with a lower environmental impact (e.g. pork or chicken), so called consumption-side strategies.

Most of the environmental impact of production of ASF is related to production of feed. One production-side strategy to reduce the environmental impact is the use of products that humans cannot or do not want to eat, such as co-products, food-waste, and biomass from marginal lands for livestock feed (referred to as 'leftover streams' in this thesis). This strategy is effective, because feeding leftover streams to livestock transforms an inedible food stream into high-quality food products, such as meat, milk, and eggs.

Two production-side strategies that use leftover streams as livestock feed were explored in this thesis: replacing soybean meal (SBM) in diets of growing pigs with either rapeseed meal (RSM) or with waste-fed larvae meal. Replacing SBM with RSM in growing-pig diets was assessed because RSM became increasingly available following an increase in bio-energy production in the EU. In this strategy, therefore, the RSM content in pig diets increased at the expense of SBM. SBM is an ingredient associated with a high environmental impact. It was expected, therefore, that replacing SBM with RSM in pig diets would lead to a decrease in the environmental impact of pork production. Replacing SBM with waste-fed larvae meal was assessed because recent developments show the environmental benefits of rearing insects as livestock feed. Insects have a low feed conversion ratio (kg feed/kg product) and

can be consumed completely, without residual materials, such as bones or feathers. The nutritional value of insects is high, especially as a protein source for livestock. Insect-based feed products, therefore, can replace conventional feed ingredients, such as SBM. Altogether this strategy suggests that waste-fed larvae meal might become an important alternative feed source in the future.

To gain insight into the status quo of the environmental impact of both mitigation strategies, replacing SBM with RSM or with waste-fed insects, we first used the attributional life cycle assessment (ALCA) method. Based on the ALCA method, results showed that each mitigation strategy was promising. Replacing SBM with RSM in growing pig diets hardly changed either global warming potential (GWP) or energy use (EU), but decreased land use (LU) up to 16% per kg body weight gain. As expected, feed production had the largest environmental impact, responsible for about 50% of the GWP, 60% of the EU, and 77% of the total LU. Feed production in combination with feed intake, were the most sensitive parameters; a small change in both these two parameters changed the results. Replacing SBM with waste-fed larvae meal in growing-pig diets showed that EU hardly changed, but GWP (29%) and LU (54%) decreased per kg body weight gain. Based on ALCA results, each mitigation strategy, therefore, seems to offer potential to reduce the environmental impact of pork production. An ALCA, however, has two disadvantages: it does not account for product-packages and it does not consider feed-food competition.

The first disadvantage of ALCA was that the complexity of dealing with product-packages is not fully considered. ‘Product-package’ refers to a multiple-output situation. During the processing of sugar beet, for example, beet-pulp and molasses are produced in addition to sugar. Sugar, beet-pulp, and molasses together form a ‘package of products’ because they cannot be produced independently from each other. An ALCA does not account for the fact that the production volume of the co-product(s) depends on the demand for the determining product (e.g. sugar), which results in the limited availability of co-products. Increasing the use of co-products in animal feed, consequently, results in reducing use of a co-product in another sector, requiring them to be replaced with a different product. The environmental impact of increasing the use of a co-product or food-waste, therefore, depends on the net environmental impact. The net environmental impact refers to the environmental benefits of using the product in its new application minus the environmental cost of replacing the product in its old application.

A consequential theoretical framework was developed to account for product-packages. The results, based on the consequential framework, contradicted standard ALCA results. The consequential LCA (CLCA) method we used for replacing SBM with RSM showed an increased GWP (up to 15%), EU (up to 12%), and LU (up to 10%) per kg body weight gain. Moreover, this CLCA method showed that replacing SBM with waste-fed larvae meal increased GWP (60%) and EU (90%), but decreased LU (73%) per kg body weight gain.

Accounting for product-packages increased the net environmental impact of each strategy, replacing SBM with RSM or with waste-fed larvae meal. The difference in results between ALCA and CLCA was especially large in the strategy with waste-fed larvae meal. The difference was caused mainly by the use of food-waste. Food-waste fed to larvae was used initially to produce bio-energy via anaerobic digestion. In CLCA, the environmental impact related to replacing the bio-energy function of food-waste with fossil-energy was included. The net environmental impact became negative, because environmental benefits of replacing SBM with waste-fed larvae meal were less than environmental costs related to the marginal energy source, i.e. fossil energy, replacing the bio-energy. Results of the indirect environmental impact, however, are situation specific: if the marginal energy source were wind or solar energy, the net environmental impact of using waste-fed larvae meal might be positive. Waste-fed larvae meal, therefore, appears to be an interesting mitigation strategy only when energy from wind and solar energy are used more dominantly than energy from fossil sources.

If results were based solely on ALCA, then these potentially negative impacts would have been overlooked. Consideration of the environmental consequences of product-packaging, therefore, is essential to determine total environmental costs. If policy makers or the feed industry want to assess the net environmental impact of a potential mitigation strategy, then we recommend to perform a CLCA instead of an ALCA. The framework developed in this thesis can be used to perform such an assessment.

The second disadvantage of an LCA was that it does not take into account feed-food competition, e.g. competition for land between humans and animals. Most LCA studies focus on the total amount of land required to produce one kg ASF. LCA studies do not account for competition for land between humans and animals, or so-called feed-food competition. In other words, they do not include, differences in the consumption of human-edible products by various livestock species or differences in the suitability of land used for feed production as land to cultivate food-crops directly. Given the global constraints on land, it is more efficient to grow food directly for human consumption rather than for livestock. To address the contribution of livestock to a future sustainable food supply, a measure for land use efficiency was developed, called the land use ratio (LUR). The LUR accounts for plant productivity, efficiency of converting human-inedible feed into ASF, and suitability of land for crop cultivation. The LUR also has a life-cycle perspective.

Results of the LUR illustrated that dairy cows on sandy soil, laying hens, and pig production systems in the Netherlands have a LUR >1.0. In terms of protein produced per m^2, therefore, it is more efficient to use these soils for livestock production to produce crops for direct human consumption than to produce feed for livestock. Only dairy cows on peat soil produce human digestible protein (HDP) more efficiently than crops do, because peat is not suitable for crop production. The LUR allows identification of livestock production systems that are able to produce HDP more efficiently than crops do. Livestock systems with a LUR<1.0, such as dairy on peat, have an important role to play in future sustainable nutrition supply.

Results of the LUR showed that livestock production systems using mainly co-products, food-waste, and biomass from marginal land, can produce human digestible protein more efficiently than crop production systems do. The availability of those leftover streams, however, is limited and, therefore, the amount of ASF produced based only on leftover streams is also limited. Because LUR is a ratio, LUR results do not give an indication of how much ASF can be produced based on livestock systems that feed mainly on leftover streams.

The third, and last, mitigation strategy, therefore, focused on the amount of ASF that can be consumed by humans, when livestock are fed only on leftover steams, also referred to as "default livestock". The calculation of the amount of ASF was based on the assumption that a vegan diet was consumed in principle. The resulting co-products and food-waste were fed to pigs and, biomass from grazing land was fed to ruminants. Results showed that in total 21 g animal source protein per person per day could be produced by feeding livestock entirely on leftovers.

Considering feed-food crops and feeding food-waste made an important contribution to the 21 g of protein that could be produced from default livestock. Considering feed-food crops implies that choices have to be made between different crops, based on their contribution to feed and food production. Oil production from soy cultivation, for example, resulted in the co-product SBM. Results showed that considering feed-food crops can affect the final protein production from pork. The practice of feeding food-waste to livestock is currently prohibited due to problems of food safety but the practice shows potential in extensively reducing the environmental impact of livestock production. Considering feed-food crops and feeding food-waste are examples of mitigation strategies that currently can be implemented to reduce further the environmental impact of the livestock sector.

On average, it is recommended to consume about 57 g of protein from ASF or plant-origin per person per day. Only ASF from default livestock does not fulfil the current global protein consumption of 32 g per person per day, but about one third of the protein each person needs

can be produced without any competition for land between feed and food production. To feed the world more sustainably, by requiring livestock production systems with a LUR <1.0, however, a paradigm shift is needed. Global average consumption of ASF should decrease to about 21 g of protein per person per day. Innovations are needed, moreover, to overcome problems of food safety and technical concerns related to collecting the leftover streams. This applies, in particular to food-waste, which is currently unused in livestock production but which presents a valuable strategy in mitigating environmental impacts caused by livestock production. Livestock systems should change their focus, furthermore, from increasing productivity per animal towards increasing protein production for humans per ha. By using leftover streams optimally, the livestock sector is able to produce a crucial amount of protein, while still avoiding competition for land between feed and food crops. Livestock, therefore, can make an important contribution to the future nutrition supply.

Samenvatting

Wereldwijd staat de productie van duurzaam voedsel hoog op de politieke agenda. Het politieke debat gaat over de uitdaging om op een duurzame manier voldoende voedsel te produceren voor een groeiende wereldbevolking. De productie van met name dierlijke producten is een grote bron van milieubelasting. De productie van dierlijke producten draagt bijvoorbeeld bij aan klimaatverandering, en aan verzuring en vermesting van onze ecosystemen. Van alle broeikasgassen die worden uitgestoten door menselijk handelen is wereldwijd ongeveer 15% afkomstig van de veehouderijsector. De veehouderijsector maakt daarnaast ook gebruik van schaarse bronnen, zoals land, water en fossiele brandstoffen. Wereldwijd wordt bijvoorbeeld 70% van al het agrarisch land gebruikt door de veehouderijsector.

Om de wereld op een duurzame manier te voeden zijn we op zoek naar mitigatiestrategieën: strategieën die de milieubelasting van de veehouderijsector verlagen. Deze mitigatiestrategieën richten zich veelal op de productie van ons voedsel, bijvoorbeeld door de milieubelasting per kg vlees te verlagen door technologische ontwikkelingen. Mitigatiestrategieën kunnen echter ook gericht zijn op veranderingen in consumptiepatronen, bijvoorbeeld minder vlees consumeren of producten consumeren met een lagere milieubelasting (zoals bijvoorbeeld kippenvlees i.p.v. rundvlees).

Wanneer we naar de milieubelasting van de productie van dierlijke producten kijken dan weten we dat het grootste deel van de milieubelasting wordt veroorzaakt door de productie van veevoer. Een manier om deze belasting te reduceren is om producten aan dieren te voeren die wij mensen niet willen of kunnen eten (reststromen genoemd). Denk hierbij aan voedselresten, bijproducten van de voedingsmiddelenindustrie of gras van marginale gronden. Voedselresten kunnen afkomstig zijn van de supermarkt, restaurants maar ook van huishoudens. Bijproducten van de voedingsmiddelenindustrie worden geproduceerd tijdens de productie van ons voedsel. Aardappelschillen worden bijvoorbeeld geproduceerd tijdens de productie van friet. Gras van marginale gronden kan worden gebruikt omdat op deze gronden de productie van humane voedsel, zoals tarwe, niet mogelijk is. Wat al deze producten gemeen hebben is dat ze niet in concurrentie zijn met de productie van voedsel voor de mens. Het gebruik van deze producten als veevoer is een effectieve strategie omdat het ervoor zorgt dat producten die niet eetbaar zijn voor de mens worden omgezet in hoogwaardige voedingsmiddelen, zoals vlees, melk en eieren.

In dit proefschrift zijn drie mitigatiestrategieën onderzocht. De eerste twee strategieën richten zich op de productiekant en onderzoeken het gebruik van restromen als veevoer. De laatste strategie combineert de productiekant met de consumptiekant. We richten ons eerst op de twee mitigatiestrategieën aan de productiekant: het vervangen van sojameel in het voer van vleesvarkens door raapzaadmeel of door meel van vliegenlarven, die gevoerd zijn op voedselresten en mest. Het vervangen van sojameel door raapzaadmeel is onderzocht omdat de beschikbaarheid van raapzaadmeel is toegenomen door de toename in productie van bio-energie in de EU. In deze strategie neemt raadzaadmeel daarom toe ten koste van sojameel. Dit is een interessante mitigatiestrategie omdat sojameel een ingrediënt is met een hoge milieubelasting. De verwachting was daarom dat het vervangen van sojameel door raapzaadmeel in varkensvoer zou leiden tot een daling van de milieubelasting van varkensvleesproductie.

Het vervangen van sojameel door larvenmeel is onderzocht omdat recente ontwikkelingen indiceren dat het kweken van insecten als veevoer leidt tot een verlaging van de milieubelasting. Dit wordt gebaseerd op het feit dat insecten een lage voederconversie (kg voer/kg product) hebben en in hun geheel kunnen worden geconsumeerd, zonder restmaterialen als botten of veren. De voedingswaarde van insecten is hoog, met name als eiwitbron voor vee. Op insecten gebaseerde producten kunnen daarom conventionele ingrediënten zoals sojameel of vismeel vervangen. Men verwacht derhalve dat op voedselresten en mest gevoerde vliegenlarven een belangrijke alternatieve voerbron kan worden in de toekomst.

Om inzicht te krijgen in de milieubelasting van beide mitigatiestrategieën (het vervangen van sojameel door raapzaadmeel of door meel van op voedselresten en mest gevoerde vliegenlarven) is een levenscyclus analyse (LCA) uitgevoerd. Er bestaan twee LCA-methodes namelijk de "attributional" LCA (ALCA) en de "consequential" LCA (CLCA). De ALCA is de meest gebruikte methode en geeft inzicht in de milieubelasting van een bepaald product, op een bepaald moment (de status quo). Met een ALCA wordt de milieubelasting, gerelateerd aan ieder proces van de productieketen van een bepaald product bij elkaar opgeteld. Op deze manier wordt er uiteindelijk één getal gegeneerd dat de milieubelasting (bijvoorbeeld de impact op klimaatverandering) van dat product (bijvoorbeeld kg varkensvlees) weergeeft.

Om inzicht te krijgen in de status quo van de mitigatiestrategieën hebben we eerst een ALCA uitgevoerd. Volgens de ALCA-methode resulteerden beide mitigatiestrategieën in een verlaging van de milieubelasting. Het vervangen van sojameel door raapzaadmeel zorgde voor een daling tot 16% in landgebruik per kg levend gewicht van het vleesvarken, maar had nauwelijks effect op de uitstoot van broeikasgassen en het energiegebruik. Zoals verwacht verklaarde de productie van voer het merendeel van de milieubelasting, namelijk 50% van de uitstoot van broeikasgassen, 60% van het energiegebruik en 77% van het totale landgebruik. Voerproductie, in combinatie met voeropname, waren de gevoeligste parameters; slechts een kleine verandering in deze twee parameters had al een effect op de uiteindelijke resultaten.

Het vervangen van sojameel door larvenmeel in het voer van vleesvarkens zorgde voor een daling van de uitstoot van broeikasgassen (29%) en het landgebruik (54%), maar had nauwelijks een effect op energiegebruik per kg levend gewicht van het vleesvarken. Gebaseerd op de resultaten van de ALCA-methode, hadden beide mitigatiestrategieën potentie om de milieubelasting van de productie van varkensvlees te verlagen.

De ALCA-methode heeft echter twee nadelen. Deze methode houdt geen rekening met de beperkte beschikbaarheid van bijproducten en voedselresten, en houdt geen rekening met de concurrentie tussen mens en dier om land. Bijproducten en voedselresten zijn onderdeel van een zogenaamd ‘product-package’ (een pakketje aan producten). Gedurende de verwerking van suikerbiet wordt naast suiker bijvoorbeeld ook bietenpulp en molasse geproduceerd. De verwerking van suikerbiet leidt dus tot verschillende eindproducten. Suiker, bietenpulp en molasse vormen samen een ‘product pakkage’ omdat ze niet onafhankelijk van elkaar kunnen worden geproduceerd. De ALCA-methode houdt geen rekening met het feit dat het volume van de bijproducten afhangt van de vraag naar het hoofdproduct (in dit geval suiker), hetgeen resulteert in een beperkte beschikbaarheid van bijproducten. De meeste bijproducten en voedselresten hebben bovendien al een bepaalde functie. Een toename in het gebruik van bijproducten in diervoeder reduceert het gebruik van dit bijproduct in een andere sector, waardoor dit vervangen dient te worden door een ander hoofdproduct. De milieubelasting van de toename in het gebruik van een bijproduct of voedselresten hangt daarom af van de netto milieubelasting. De netto milieubelasting is het resultaat van de verandering in belastingdoor het gebruiken van het bijproduct in de nieuwe toepassing min de verandering in belasting door het vervangen van het bijproduct in de oude toepassing.

De CLCA-methode kan hier wel rekening mee houden. De CLCA-methode kijkt namelijk niet alleen naar de milieubelasting van het product zelf maar ook naar de effecten buiten de productieketen. Het feit dat bijproducten beperkt beschikbaar zijn en al een andere toepassing kunnen hebben zou dus naar voren moeten komen als een consequentie. De CLCA-methode is echter nog een relatief nieuwe methode die nog in ontwikkeling is. De CLCA-methode heeft nog geen richtlijnen over het omgaan met bijproducten. In dit proefschrift is daarom een raamwerk ontwikkeld dat richting geeft aan omgaan met bijproducten en voedselresten op basis van de CLCA-methode. Dit raamwerk is eerst geïllustreerd met diverse casestudies en vervolgens toegepast op de twee mitigatiestrategieën (het vervangen van sojameel door raapzaadmeel of door larvenmeel). Dit leverde tegenstrijdige resultaten op ten opzichte van de ALCA-resultaten. De CLCA-resultaten lieten zien dat het vervangen van sojameel door raapzaadmeel resulteerde in een toename van 15% van de uitstoot van broeikasgassen, een toename tot 12% energiegebruik en een toename van 10% landgebruik per kg levend gewicht. Het vervangen van sojameel door larvenmeel resulteerde in een 60% toename van de uitstoot van broeikasgassen, 90% toename in energiegebruik, maar een 73% afname in landgebruik per kg levend gewicht. Als we dus rekening houden met de consequenties van het gebruik van bijproducten en voedselresten dan leveren deze twee mitigatiestrategieën geen milieuwinst op. Tussen de twee methodes, ALCA en CLCA, zien we met name een groot verschil in resultaten in de situatie van larvenmeel. Dit verschil werd voornamelijk veroorzaakt door het gebruik van voedselresten. De voedselresten die worden gevoerd aan de vliegenlarven worden in eerste instantie gebruikt voor de productie van bio-energie via anaerobe vertering. Omdat de voedselresten maar beperkt beschikbaar zijn, kunnen de voedselresten nu niet meer worden gebruikt om bio-energie te produceren. De bio-energie, die nu niet meer geproduceerd wordt, moet worden vervangen, in dit geval door fossiele energie. Fossiele energie heeft een hoge milieubelasting. De netto milieubelasting werd negatief omdat het milieuvoordeel van het vervangen van sojameel door larvenmeel minder was dan de milieubelasting gerelateerd aan de toename in gebruik van fossiele brandstoffen. De resultaten van de indirecte milieubelasting (CLCA) zijn echter situatie afhankelijk: indien men geen fossiele brandstoffen zou gebruiken om de bio-energie te vervangen maar bijvoorbeeld wind- of zonne-energie dan zou de netto milieubelasting van het gebruik van larvenmeel positief zijn. Fossiele brandstoffen worden echter het meest gebruikt en een daling in de productie van bio-energie zal daarom hoogstwaarschijnlijk leiden tot een stijging van de productie van fossiele brandstoffen. Larvenmeel gevoerd op voedselresten lijkt daarom alleen een interessante mitigatiestrategie als wind- en zonne-energie meer worden gebruikt dan fossiele bronnen.

De resultaten van de twee mitigatiestrategieën laten het belang van de CLCA-methode zien. Als de resultaten alleen op de ALCA-methode zouden zijn gebaseerd dan zouden de negatieve milieueffecten van de indirecte consequenties over het hoofd zijn gezien. Het overwegen van de indirecte consequenties is daarom essentieel om de milieubelasting van een strategie te bepalen. Indien beleidsmakers of de voedingsindustrie de netto milieubelasting van een potentiele mitigatiestrategie willen bepalen raden we aan de CLCA-methode te gebruiken in plaats van de ALCA-methode. Het raamwerk dat in dit proefschrift is ontwikkeld kan worden gebruikt bij een dergelijk beoordeling.

Het tweede nadeel van de ALCA-methode is dat deze geen rekening houdt met de concurrentie tussen mens en dier om land. Omdat land schaars is vindt er concurrentie plaats tussen productie van veevoer en productie van voedsel. De meeste ALCA-studies richten zich op het totaal aan land dat nodig is om een kg dierlijk product te produceren. De ALCA-methode houdt op die manier echter geen rekening met de concurrentie tussen mens en dier. Rekening houdend met de schaarste aan land, is het efficiënter om gewassen te verbouwen die geschikt zijn voor humane consumptie i.p.v. het land te gebruiken om gewassen te verbouwen om veevoer te produceren. Om de bijdrage van de veehouderijsector aan een - in de toekomst duurzame - voedselvoorziening in kaart te brengen hebben we een methode ontwikkeld, ook wel de landgebruikratio (in Engels 'land use ratio oftewel LUR) genoemd. De LUR houdt rekening met de productiviteit van een gewas, het gebruik van voor de mens niet eetbare producten als veevoer en of het land geschikt is voor akkerbouw (wel of geen marginale gronden). Door hier rekening mee te houden krijg je inzicht in de optimale eiwit productie voor humane consumptie per ha. De resultaten laten zien dat melkkoeien op zandgrond, leghennen, en varkensproductiesystemen in Nederland een LUR >1 hebben. In termen van eiwitproductie per m^2 is het daarom efficiënter om deze gronden te gebruiken voor de productie van humaan voedsel in plaats van voor de productie van veevoer. Melkkoeien op veengronden hebben echter een LUR <1. Dit komt omdat veengrond niet geschikt is voor akkerbouw. Veehouderijsystemen met een LUR van <1, zoals melkkoeien op veengrond, spelen daarom een belangrijke rol in een toekomstige duurzame voedselvoorziening. De resultaten van de LUR laten zien dat veehouderijsystemen die gebruik maken van bijproducten, voedselresten en biomassa van marginale gronden efficiënt eiwit kunnen produceren. De beschikbaarheid van deze reststromen is echter beperkt en daarom is de hoeveelheid dierlijke product die geproduceerd kan worden van vee gevoerd op reststromen ook beperkt. Omdat de LUR een ratio is, geven de resultaten geen indicatie van de hoeveelheid dierlijk product dat kan worden geproduceerd door veehouderijsystemen die hoofdzakelijk reststromen gebruiken.

De derde, en laatste, mitigatiestrategie onderzocht in dit proefschrift richt zich daarom op de hoeveelheid dierlijk product dat kan worden geconsumeerd door mensen wanneer vee alleen wordt gevoerd met reststromen. Voor de berekening van de hoeveelheid dierlijk product is de aanname gedaan dat men een veganistisch dieet eet. De resulterende bijproducten en voedselresten worden aan varkens gevoerd en biomassa van grasland wordt aan herkauwers gevoerd. De resultaten laten zien dat er in totaal 21 g dierlijk eiwit per person per dag kan worden geproduceerd als vee alleen met reststromen wordt gevoerd. Dat dit getal zo hoog is kan allereerst worden verklaard door het gebruik van voedselresten. Deze voedselresten zijn hoogwaardige producten zoals bijvoorbeeld sojaolie, waarop het varken goed kan groeien.

Een tweede reden is dat sommige bijproducten ook hoogwaardige veevoerproducten zijn. Tijdens de productie van sojaolie wordt bijvoorbeeld sojameel geproduceerd. Sojameel is een heel voedingsrijk product voor varkens. Indien hier een andere aanname zou worden gedaan dan zou dit de resultaten aanzienlijk veranderen. Wanneer men bijvoorbeeld zonnebloemolie zou consumeren dan resulteert dit in het veel minder hoogwaardige bijproduct zonnebloemschroot. Dit laat dus het belang zien van het overwegen van gewassen die zowel een hoogwaardig product voor humane consumptie als een hoogwaardig bijproduct voor de veehouderij opleveren. Wanneer men streeft naar reductie van landgebruik dan dient men naar de complete 'product-package' te kijken en rekening te houden met de functies die het kan hebben voor zowel humane consumptie als veevoer. Dit houdt dus in dat er keuzes moeten worden gemaakt tussen gewassen op basis van hun bijdrage aan de productie van voedsel en voer.

Het voeren van voedselresten is tot nu toe verboden in verband met voedselveiligheid, maar het kan potentieel bijdragen aan een daling van de milieubelasting van de veehouderijsector. Het overwegen van gewassen die bijdragen aan de productie van zowel voedsel als veevoer en het voeren van voedselresten zijn voorbeelden van mitigatiestrategieën die op dit moment kunnen worden geïmplementeerd om de milieubelasting van de veehouderijsector in de toekomst te beperken.

Het wordt aangeraden om per person per dag 57 g eiwit uit dieren of planten te consumeren. De 21 g dierlijk eiwit per person per dag die kan worden geproduceerd als vee alleen met reststromen wordt gevoerd kan dus bijdragen aan ongeveer een derde van de eiwitten die we dagelijks nodig hebben. Om de wereld op een duurzamere manier te voeden en een ontwikkeling in gang te zetten naar veehouderijsystemen met een LUR <1, is echter een paradigmaverschuiving nodig. De gemiddelde wereldconsumptie van dierlijk eiwit zou moeten dalen van 32 g naar 21 g eiwit per person per dag. Daarnaast zijn innovaties nodig om de problemen met betrekking tot voedselveiligheid en technische problemen gerelateerd aan de verzameling van voedselresten op te lossen. Veehouderijsystemen zouden zich niet moeten richten op de hoogste productiviteit per dier, maar op de hoogste eiwitproductie voor mensen per hectare. Door reststromen optimaal te gebruiken kan de veehouderijsector een cruciale hoeveelheid eiwit produceren zonder dat er concurrentie om land plaatsvindt tussen voer en voedsel. De veehouderijsector kan daarom een belangrijke bijdrage leveren aan de wereld voedselproductie van de toekomst.

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About the author

Hannah van Zanten was born in Rotterdam in 1983. She obtained her BSc 'Animal Management' with a specialisation in 'Wildlife Management' at van Hall Larenstein in 2007. During her BSc study she spent one year to work as a field guide in the Kruger national Park in South Africa. She graduated cum laude from Wageningen University in 2009 with a master degree in Animal Sciences. Her first thesis was carried out within Public Administration and Policy Group and focused on the interaction between the media and the members of the Dutch parliament related to the topic animal welfare. Her second thesis was carried out within the Animal Production Systems group and focused on the environmental and welfare benefits of improved climate conditions in broiler stables, due to the use of a heat exchanger, (resulting in a scientific publication). After graduation she started the honours programme 'Rijkstraineeprogramma' of the Dutch government and worked as a policy maker within the ministry of Economic Affairs in the Hague for two years. Next, she started as a PhD student within the Animal Production Systems group of Wageningen University in 2011. Her PhD study focussed on the environmental benefits of using human-inedible-sources as livestock feed. Hannah won the Storm-van der Chijs Stipend in 2015, which is awarded to the three most talented female PhD candidates of Wageningen University. Furthermore, she received the award for the best oral presentation at the EAAP (European Association of Animal Production) in Copenhagen in 2014, and the award for the best oral presentation at the WIAS Science day in 2016.

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Sikkema A, Van Zanten HHE, (2015) Vlieg als veevoer kost meer CO_2 dan soja. Resource: weekblad voor Wageningen UR 9 (21) p. 10

Education certificate

Completed training and supervision plan[1]

The Basic Package (1.5 ECTS)

International conferences (4.9 ECTS)
Annual Meeting of the European Federation of Animal Science, Nantes, 26-30 August 2013
Annual Meeting of the European Federation of Animal Science, Copenhagen, 25-29 August 2014
Annual Meeting of the European Federation of Animal Science, Warsaw, 31 August-4 September 2015
Conference Insects to feed the world, Netherlands, 14-17 May 2014

Seminars and workshops (2.7 ECTS)
WIAS Science Day, Wageningen (2013 - 2016)
Symposium 'Innovation born of Integration' (2012)
Sustainable intensification of smallholder livestock production: fact and fiction (2013)
Symposium "Fibres in food and feed" (2013)
EU workshop 'TRANSMANGO' (2015)
EU workshop 'SUSFANS' (2015)

Presentations (7 ECTS)
Poster presentation WIAS Science Day, Wageningen (2014)
Oral presentation Annual Meeting of the European Federation of Animal Science (invited speaker) (2013)
Oral presentation conference Insects to feed the world (2014)
Oral presentation Annual Meeting of the European Federation of Animal Science (2014)
Oral presentation Annual Meeting of the European Federation of Animal Science (2015)
Oral presentation Fibl, Switzerland (2015)
Oral presentation WIAS Science Day, Wageningen (2016)

In-Depth Studies (6.6 ECTS)
Course Simapro (2011)
Course 'Pig Feed', Wageningen University (2012)
PhD Course Environmental Impact Assessment of Livestock Systems, Wageningen University (2015)
Course Advances in Feed Evaluation Science, Wageningen University (2015)
LCA discussion group

Professional Skills Support Courses (3 ECTS)
High-Impact Writing in Science, Wageningen University (2013)
Supervising MSc thesis work, Wageningen University (2013)
Giving lectures, Wageningen University (2015)
Designing exams, Wageningen University (2015)

Research Skills Training (8.6 ECTS)
Preparing PhD research proposal
Review scientific papers
Supervision PhD student

Lecturing (4.5 ECTS)
BSc course Global and Sustainable Animal Production in the 21ste century (2015)
MSc course (APS) Issues and Options (2013)
MSc course Research Master Cluster (2015)
PhD course Environmental Impact Assessment of Livestock Systems (2015)

Supervising practicals and excursions (5.8 ECTS)
BSc course Inleiding Dierwetenschappen (farm visits) (2012)
BSc course Global and Sustainable Animal Production in the 21ste century (2015)
MSc course (APS) Issues and Options (Essays and practical) (2013 + 2015)
MSc course Research Master Cluster (review proposals) (2014)

Supervising MSc theses (14.5 ECTS)

Preparing course material (4.9 ECTS)
MSc course (APS) Issues and Options: consequential LCA lecture and practical (2013)
BSc course global and Sustainable Animal Production in the 21ste century: writing course material (2015)
PhD course: consequential LCA lecture and practical (2015)

Organisation of seminars and courses (1.5 ECTS)
Co-ordinator LCA discussion group 2012-2013
EU workshop SUSFANS 2015

Total: 66.5 ECTS

[1]With the activities listed the PhD candidate has complied with the educational requirement set by the Graduated School of Wageningen University Institute of Animal Sciences (WIAS). One ECTS equals a study load of 28 hours.

Colophon

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