Milk Production & Greenhouse Gases

Integrated modeling of feeding and breeding strategies to reduce emissions

Corina van Middelaar

Thesis committee

Promotor

Prof. Dr I.J.M. de Boer
Professor of Animal Production Systems
Wageningen University

Co-promotors

Dr P.B.M. Berentsen
Associate professor, Business Economics Group
Wageningen University

Dr J. Dijkstra
Associate professor, Animal Nutrition Group
Wageningen University

Other members

Dr A. Bannink, Wageningen UR Livestock Research, Lelystad

Prof. Dr O. Oenema, Wageningen University

Dr E. Wall, Scotland's Rural College, Midlothian, Scotland

Dr A.G. Williams, Cranfield University, Bedford, United Kingdom

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Corina van Middelaar

Thesis

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Abstract

Dairy cattle are responsible for about 30% of global greenhouse gas (GHG) emissions produced by the livestock sector. Main sources of emissions are enteric fermentation (methane), feed production (mainly carbon dioxide and nitrous oxide), and manure management (methane and nitrous oxide). The research described in this thesis aims to develop an integrated method to evaluate strategies to reduce GHG emissions from dairy production at the chain level, and to evaluate feeding and breeding strategies using this integrated method. We first explored consequences of differences in methods and data to calculate emissions from feed production, and decided upon a standard life cycle assessment (LCA). Subsequently, we integrated a whole-farm optimization model with an LCA of purchased inputs, and a mechanistic model to predict enteric methane production. We used this integrated method to evaluate the impact of several feeding and breeding strategies on GHG emissions at chain level and on labor income at farm level. All strategies were evaluated for the case-study of a typical Dutch dairy farm on sandy soil. The relevance of integrated modeling was demonstrated by evaluating the impact of increasing maize silage at the expense of grass and grass silage in a dairy cow's diet at animal, farm, and chain levels. At animal level, the strategy results in an immediate reduction in GHG emissions. At farm and chain levels, it takes more than 60 years before annual emission reduction has paid off emissions from land use change. Results confirmed the importance of integrated modeling. Subsequently, other feeding strategies were evaluated, including dietary supplementation of extruded linseed, dietary supplementation of nitrate, and reducing the maturity stage of grass and grass silage. Each feeding strategy reduced GHG emissions along the chain. Supplementing diets with nitrate resulted in the greatest reduction, but reducing grass maturity was most cost-effective (i.e. lowest costs per ton CO_2-equivalents reduced). In case of breeding, two methods were explored to determine the relative importance of individual traits to reduce GHG emissions along the chain (i.e. the relative GHG value): the first method aims at maximizing labor income, the second at minimizing GHG emissions per kg milk. GHG values were calculated for one genetic standard deviation change of milk yield and longevity, while robustness of results was explored by comparing GHG values for an efficient and a less-efficient farm. The GHG values of both milk yield and longevity were at least twice as great when focus was on minimizing GHG emissions. Furthermore, the GHG value of milk yield was greater than that of longevity when focus was on maximizing labor income, especially for the less efficient farm. When focus was on minimizing GHG emissions, both traits were equally important on each level of efficiency. To substantially reduce GHG emissions from dairy production, a combination of strategies is required.

Contents

Chapter 1

General introduction

1 Background

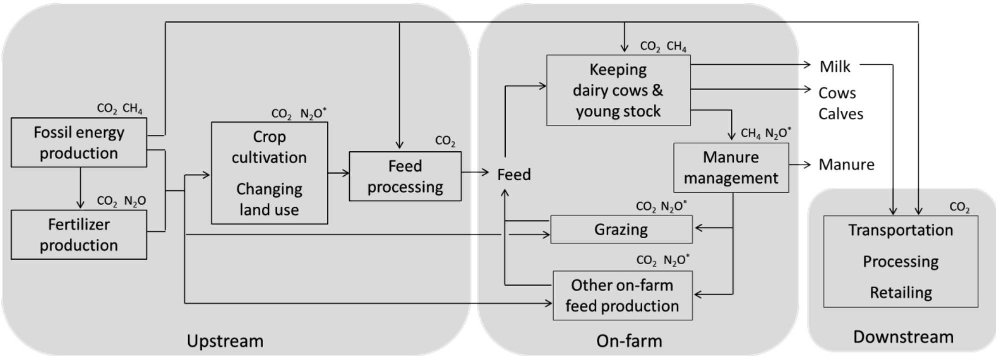
Emissions of greenhouse gases (GHG) from human activities are likely to contribute to climate change (IPCC, 2007). Climate change has been related to rising sea levels, extreme weather conditions, air pollution, and loss of biodiversity, among other effects (Walther et al., 2002; McMichael et al., 2006). Such effects can damage ecosystems and human health. To monitor GHG emissions from human activities, initiatives to calculate and report GHG emissions of products and services from humans have increased (Muñoz et al., 2013).

A recent report from the Food and Agricultural Organization (Gerber et al., 2013) demonstrates that the livestock sector is one of the main contributors to GHG emissions induced by human activities. The sector is responsible for about 7.1 gigatonnes CO_2 equivalents (CO_2e) per year (based on 2005), which equals 14.5% of total emissions induced by humans. The demand for animal-source food is expected to double by 2050 because of growth of the world population, increase in incomes, and urbanization (Rae, 1998; FAO, 2009). Identification and application of strategies to reduce GHG emissions from livestock systems, therefore, is important (Smith et al., 2007).

1.1 GHG emissions from dairy production

Dairy cattle, producing milk, meat, and non-edible products (e.g. manure), are responsible for about 30% of global GHG emissions produced by the livestock sector (Gerber et al., 2013). Main pathways of GHG emissions from dairy production in developed countries are shown in Figure 1. Processes along the dairy production chain include processes related to production of farm-inputs (upstream), processes related to on-farm milk production (on-farm), and processes related to transport and processing of milk (downstream). Important GHG emissions from dairy production are carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O).

Emission of CO_2 results from combustion of fossil fuels to power machinery and from burning and microbial decay of biomass related to, for example, changes in land use or crop management. Emission of CO_2 also results from animal respiration. These emissions, however, are not included in GHG calculations, because they are assumed to be balanced by uptake of CO_2 by plants consumed by the animal (Rypdal et al., 2006). Emission of CH_4 results from decomposition of organic matter in oxygen-deprived conditions during, for example, enteric fermentation and manure management. Moreover, CH_4 is emitted during fossil fuel extraction and refining.



*Including direct and indirect N_2O emissions. Indirect N_2O emission results from release of NH_3, NO_X, and NO_3^-

Figure 1. Main pathways of GHG emissions from dairy production in developed countries.

Emission of N_2O results from microbial transformation of nitrogen in the soil or in manure (nitrification in combination with incomplete denitrification), as well as from production of nitrogen fertilizer. Two types of N_2O emission are defined: direct and indirect emission. Direct N_2O emission results from manure storages and nitrogen application to the field (e.g. fertilizers, crop residues) during crop cultivation, from changes in land use or crop management, and from industrial processes. Indirect N_2O emission results from microbial transformation of nitrogen released into the environment as ammonia (NH_3), nitrogen oxide (NO_x), or nitrate (NO_3^-). In addition to the emission of GHGs, dairy production can contribute to CO_2 sequestration, because grassland soils are an important carbon sink (Soussana et al., 2010).

Contributions of different processes to total GHG emissions from dairy production are shown in Figure 2. Results are based on Van Middelaar et al. (2011) and Gerber et al. (2013), and apply to dairy production in developed countries. Different GHG emissions were summed up based on their equivalence factor in terms of CO_2 equivalents (100-year time horizon): 1 for CO_2, 25 for CH_4, and 298 for N_2O (Forster et al., 2007). Emissions from manure management include those from manure in stable and storages, and those from manure during grazing. Emissions from manure application during crop cultivation are allocated to feed production. Emissions from feed production include those from both on-farm and off-farm feed production, and from changes in land use. Off-farm feed production involves cultivation and processing of purchased feed products. Emissions from changes in land use were limited to those from deforestation related to the expansion of soybean production in Brazil and Argentina (Gerber et al., 2013). Emissions from energy sources include production and combustion of energy used during on-farm processes, except for feed production. Downstream processes were limited to transport and processing of the milk, up to the retailer.

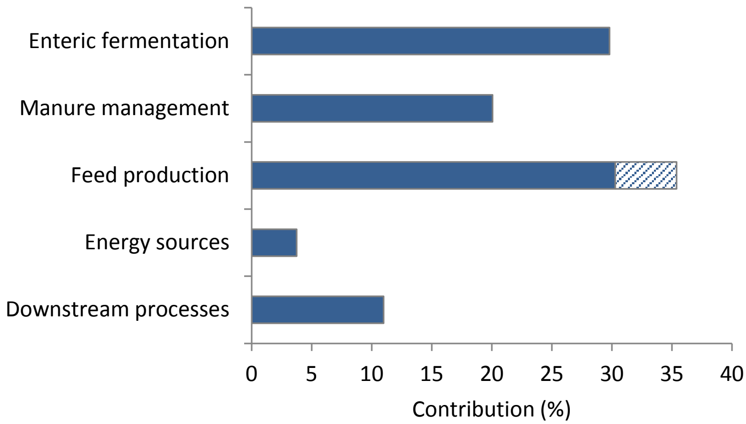


Figure 2. Contributions of different processes to total GHG emissions from dairy production in developed countries. The pattern fill shows the contribution of changes in land use for feed production.

Main sources of GHG emissions from dairy production are enteric fermentation (CH_4), feed production (mainly CO_2 and N_2O), and manure management (CH_4 and N_2O). Enteric fermentation and feed production each contribute about 30% to total emissions, whereas manure management contributes about 20%. Including emissions from changes in land use (mainly CO_2) increases the contribution of feed production. Production and combustion of energy sources used during on-farm processes contribute only about 4% to total emissions, whereas energy used during downstream processes contribute about 11%.

1.2 Strategies to reduce GHG emissions

Various strategies have been proposed to reduce GHG emissions from dairy production (De Boer et al., 2011). Most strategies apply to upstream and on-farm processes (i.e. including the three main sources of GHG emissions), and originate from specialized disciplines, such as animal feeding, plant or animal breeding, or manure processing technology (e.g. Ellis et al., 2008; Wall et al., 2010; De Vries et al., 2012). From a perspective of animal sciences, important areas of interest to reduce GHG emissions per kg milk are feeding strategies to reduce emissions from enteric fermentation and feed production, and breeding strategies to improve animal productivity. Because climate change is a global issue, strategies to reduce GHG emissions from dairy production in developed countries should aim at reducing emissions per unit of product (e.g. per kg milk), rather than reducing emissions per cow or per ha of land.

Feeding strategies to reduce GHG emissions from dairy production focus mainly on the emission of enteric CH_4 (Kebreab et al., 2006a; Grainger and Beauchemin, 2011). Enteric CH_4 derives from microbial fermentation of feed substrates in the rumen (92%) and large intestine (8%) (Bannink et al., 2011). Fermentation of structural carbohydrates, such as cellulose and hemicellulose, generally increase CH_4 production, whereas fermentation of non-structural carbohydrates, such as starch, generally decrease CH_4 production. Feeding strategies to reduce enteric CH_4 include supplementing diets with fatty acids, increasing the amount of concentrates in the diet, replacing grass silage with maize silage, and improving roughage quality (Glasser et al., 2008; Dijkstra et al., 2011; Brask et al., 2013). In addition, application of precision feeding (i.e. balancing feed intake with feed requirements) can contribute to reducing GHG emissions from dairy production by reducing the amount of feed per kg milk produced, i.e. reducing enteric CH_4 production as well as emissions from feed production. A final feeding strategy to reduce GHG emissions is the use of feed products with a low environmental impact, such as by-products.

Selective breeding for increased milk yield, reduced calving interval, and increased longevity are examples of breeding strategies to improve animal productivity and reduce GHG emissions per kg milk (Wall et al., 2010; Bell et al., 2011). By increasing milk yield per cow (i.e. fat-and-protein corrected milk) from 6,270 kg/year in 1990 to 8,350 kg/year in 2008, for example, enteric CH_4 production reduced from 17.6 to 15.4 g per kg milk (Bannink et al., 2011). Reducing calving interval increases the average daily milk yield per cow and reduces involuntary culling, whereas increasing longevity reduces the number of replacement heifers. An improvement of these traits increases life-time milk yield per cow and decreases the number of non-productive animals in the herd, both contributing to a reduction in GHG emission per kg milk. Other examples of breeding strategies to reduce GHG emissions from dairy production are selective breeding for improved feed efficiency or for reduced CH_4 production (Basarab et al., 2013).

2 Knowledge gaps

Most studies that evaluated the impact of feeding and breeding strategies to reduce GHG emissions from dairy production focused only on the emission of enteric CH_4 (Grainger and Beauchemin, 2011; Basarab et al., 2013). The advantage of such studies is that they provide a great understanding of the impact of the strategy at the level of the animal. Implementing a feeding or breeding strategy at a commercial farm, however, affects not only the animal, but also other aspects related to dairy farming, such as on-farm feed production. As a result, not only the emission of enteric CH_4, but also other GHG emissions might change (Chianese et al., 2009). In addition, a strategy can affect the type and amount of purchased products, such as feed and fertilizers (Williams et al., 2014). Hence, GHG emissions related to upstream processes might change as well. Finally, a strategy can affect downstream processes, because of, for example, changes in the ratio of milk to meat production (Zehetmeier et al., 2012). To understand which strategies can contribute to reducing the net contribution of dairy production to global GHG emissions, therefore, an integrated approach is required that accounts for all changes in farm management and includes all changes in GHG emissions along the chain.

Life cycle assessment (LCA) is a scientifically accepted and internationally standardized method to evaluate use of resources and emission of pollutants along an entire production chain (ISO 14040 and ISO 14044). LCA, therefore, can be used to calculate CO_2, CH_4, and N_2O emissions from dairy production, including upstream, on-farm, and downstream processes. To evaluate the impact of strategies to reduce GHG emissions by means of LCA, however, some aspects should be taken into consideration.

First, methods and data to calculate emissions from feed production are highly variable, and data can be subject to high uncertainty (Flysjö et al., 2011a). Accounting for GHG emissions and carbon sequestration from changes in land use and crop management, in particular, appears complex (Flysjö et al., 2012). Insight into the impact of differences in methods and data on GHG emissions per kg feed is lacking.

Second, most LCA studies on GHG emissions from dairy production use empirical methods to calculate the emission of enteric CH_4 (e.g. Thomassen et al., 2008). Empirical methods can be used to gain insight into the average amount of enteric CH_4 production per cow, but such methods are less suitable to evaluate the impact of dietary changes. To evaluate the impact of dietary changes, mechanistic modeling of enteric fermentation is required. Mechanistic models are found to be more precise than empirical methods and, therefore, provide a better alternative when evaluating the impact of strategies that influence the diet (Alemu et al., 2011).

Third, LCA does not provide insight into consequences of a strategy on farm management, such as changes in diets, in on-farm feed production, or in purchases of feed products and fertilizers. To simulate the consequences of a strategy on farm management, whole-farm modeling techniques are required (Berentsen and Giesen, 1995; Schils et al., 2007). An integrated method that combines LCA with mechanistic modeling of enteric CH_4 production and a whole-farm model to simulate changes in farm management is lacking.

Strategies to reduce GHG emissions from dairy production not only should reduce emissions along the chain, but also should be economically viable (Hristov et al., 2013a). For feeding strategies, little information is available on their impact on labor income of the farm family. For breeding strategies, economic effects of an improvement in genetic traits (i.e. economic values) are generally used to determine the relative weight for each trait in the breeding goal (Groen, 1988; Koenen et al., 2000). Methods to calculate the relative value of genetic traits to reduce GHG emissions along the chain, however, are not available.

The two objectives of this thesis are to:

- Develop an integrated method to evaluate strategies to reduce GHG emissions from dairy production at the chain level.
- Evaluate feeding and breeding strategies to reduce GHG emissions by using this integrated method.

Feeding and breeding strategies to reduce GHG emissions along the chain were evaluated in a case-study of a typical Dutch dairy farm on sandy soil.

3 Outline of the thesis

The structure of the work and chapters included in the thesis are shown in Figure 3. Results of Chapter 2 and 3 form the basis of the entire thesis: Chapter 2 evaluates methods and data to calculate GHG emissions from feed production, whereas Chapter 3 demonstrates the relevance of integrated modeling to evaluate strategies to reduce GHG emissions. Chapter 4 evaluates the impact of feeding strategies on GHG emissions and labor income of the farm family, whereas Chapter 5 presents two methods to evaluate the impact of breeding strategies. Finally, Chapter 6 explores the robustness of the methods presented in Chapter 5.

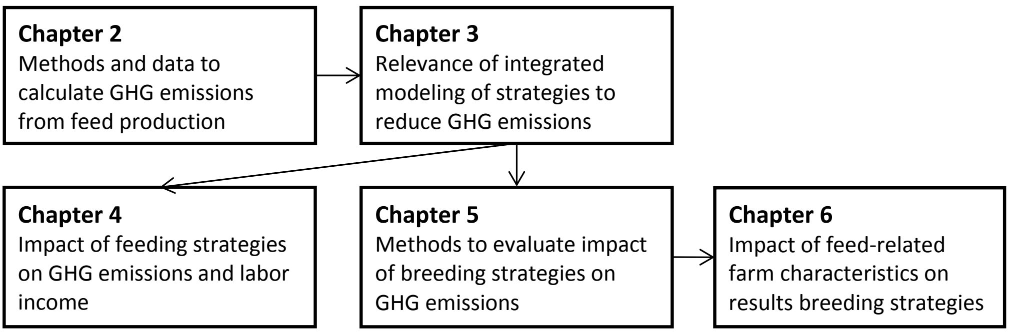


Figure 3. Structure of the work and chapters included in the thesis.

Chapter 2 Evaluates the impact of differences in methods and data to calculate GHG emissions from feed production, including emissions related to changes in crop management and land use.

Chapter 3 Demonstrates the relevance of integrated modeling by evaluating the impact of replacing grass silage with maize silage in a dairy cow's diet at animal, farm, and chain level. A whole-farm model (Berentsen and Giesen, 1995) based on the objective to maximize labor income is combined with LCA (ISO 14040 and ISO 14044) and with a mechanistic model to predict enteric CH_4 production (Dijkstra et al., 1992).

Chapter 4 Evaluates the impact of several feeding strategies on GHG emissions along the chain. In addition, the impact on labor income of the farm family is determined. Combining both impacts results in a figure representing the cost-effectiveness of the strategies. Feeding strategies under study are supplementing diets with extruded linseed, supplementing diets with nitrate, and reducing the maturity stage of grass and grass silage.

Chapter 5 Explores two methods to evaluate GHG values of genetic traits in dairy cows. Both methods, based on the same principle that is used to calculate relative economic values, determine the impact of one unit change in individual traits on GHG emissions along the chain. The first method is based on the objective to maximize labor income, the second on minimizing GHG emissions per kg milk. Economic consequences of a change in traits are taken into account in each method. Genetic traits under study are milk yield and longevity.

Chapter 6 Explores the impact of feed-related farm characteristics on GHG values of genetic traits, by comparing the values of milk yield and longevity for an efficient farm (Chapter 5) with those for a less efficient farm. The less efficient farm uses safety margins for feeding protein, and has a lower grass and maize yield per ha than the efficient farm modeled in Chapter 5.

Chapter 7 Discusses the relevance and methodological challenges of integrated modeling and places the results of this thesis in a wider context. In addition, some practical implications for reducing GHG emissions on commercial farms are discussed, and an overview of the conclusions from this thesis is given.

Chapter 2

Exploring variability in methods and data sensitivity in carbon footprints of feed ingredients

C.E. van Middelaar[a], C. Cederberg[b], T.V. Vellinga[c], H.M.G. van der Werf[d,e], and I.J.M. de Boer[a]

[a] Animal Production Systems group, Wageningen University, Wageningen, the Netherlands

[b] SIK-The Swedish Institute for Food and Biotechnology, Gothenburg, Sweden

[c] Wageningen UR Livestock Research, Animal Science group, Lelystad, the Netherlands

[d] INRA, Rennes, France

[e] Agrocampus Ouest, Rennes, France

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Abstract

Production of feed is an important contributor to life cycle greenhouse gas (GHG) emissions, or carbon footprints (CFPs), of livestock products. Consequences of methodological choices and data sensitivity on CFPs of feed ingredients were explored to improve comparison and interpretation of CFP studies. Methods and data for emissions from cultivation and processing, land use (LU), and land use change (LUC) were analyzed. For six ingredients (maize, wheat, palm kernel expeller, rapeseed meal, soybean meal and beet pulp) CFPs resulting from a single change in methods and data were compared with a reference CFP, i.e. based on IPCC Tier 1 methods, and data from literature. Results show that using more detailed methods to compute N_2O emissions from cultivation hardly affected reference CFPs, except for methods to determine NO_3^- leaching (contributing to indirect N_2O emissions) in which the influence is about -7 to +12%. Overall, CFPs appeared most sensitive to changes in crop yield and applied synthetic fertilizer-N. The inclusion of LULUC emissions can change CFPs considerably, i.e. up to 877%. The level of LUC emissions per feed ingredient highly depends on the method chosen, as well as on assumptions on area of LUC, C-stock levels (mainly above ground-C, and soil-C), and amortization period. We concluded that variability in methods and data can significantly affect CFPs of feed ingredients, and hence CFPs of livestock products. Transparency in methods and data are therefore required. For harmonization, focus should be on methods to calculate NO_3^- leaching, and emissions from LULUC. It is important to consider LUC in CFP studies of food, feed and bioenergy products.

1 Introduction

Environmental consequences of livestock production have received increasing attention over the last few years. Global warming, induced by emission of greenhouse gases (GHGs), is one of the main problems addressed (Steinfeld et al. 2006). Livestock production contributes to global warming by emission of carbon dioxide (CO_2) from fossil fuel combustion and land use change (mainly deforestation), emission of methane (CH_4) from manure and enteric fermentation by ruminants, and emission of nitrous oxide (N_2O) from manure storages and application of fertilizer for cultivation (Steinfeld et al. 2006; IPCC 2007; De Vries and De Boer 2010). With livestock production being an important contributor to GHG emissions and the growing societal concern about global warming, GHG emissions from livestock production have become an imperative study object (Ellis et al. 2008; De Boer et al. 2011).

Life cycle assessment (LCA) is an internationally accepted and standardized method (ISO 14043, 2000) to evaluate GHG emissions of a product or production system. It evaluates the use of natural resources, and emission of pollutants along the entire life cycle of a product (Guinee et al. 2002; Rebitzer et al. 2004). Carbon footprint (CFP) assessment is a single issue LCA focussing on emission of GHGs.

The CFP of various livestock products has been calculated, e.g. for milk (Haas et al. 2001; Thomassen et al. 2008; 2009; Van der Werf et al. 2009; Flysjö et al. 2011a;2012), pork (Basset-Mens and Van der Werf 2005), beef (Casey and Holden 2005; Beauchemin et al. 2011), chicken (Pelletier 2008), and eggs (Mollenhorst el al. 2006; Dekker et al. 2011). Such CFP assessments result in the identification of hotspots for GHG emissions along the production chain (Thomassen et al. 2009). A hotspot is a production stage with a high contribution to the environmental impact of a product. For most livestock products, this hotspot is feed production, including cultivation, processing, and transport stages. For milk, for example, production of feed explains around 45% of the CFP (Thomassen et al. 2008; Van Middelaar et al. 2011), for pork it is 60% and for chicken even 80% (Basset-Mens and Van der Werf 2005; Pelletier 2008). Correct assessment of the CFP of feed ingredients, therefore, is an important aspect of CFP assessment of livestock products.

To assess the CFP of feed ingredients, we need a harmonized method to calculate GHG emissions along the feed production chain. Variability in methods hampers comparison of CFP results among studies (De Vries and De Boer 2010). Particularly, accounting for emissions or C-sequestration from land use (LU) and land use change (LUC) appears complex. So far, there is no international consensus on a method to account for this, which increases variability in CFP studies (Prudencio da Silva et al. 2010; Cederberg et al. 2011; Flysjö et al. 2012). Exploring variability in methods contributes to harmonization as it identifies the aspects that lead to differences between CFP studies.

In addition to a harmonized method to calculate emissions, we need high-quality inventory data for each activity in the production chain, i.e. data on use of resources, emission of pollutants, and technical in- and outputs. Such inventory data can be subject to high uncertainty and variability (Flysjö et al. 2011a). To improve LCA studies, insight into the relation between input data and the outcome of the study is required (Steen, 1997; Sakai and Yokoyama, 2002). A sensitivity analysis shows for which data the outcome (e.g. the CFP of a product) is most sensitive. In other words, it shows which data should be considered first to improve the accuracy of an LCA study (Steen, 1997).

To improve comparison and interpretation of CFP studies, this study explored the effect of variability in methods and data sensitivity on CFPs of feed ingredients. We included emissions related to crop production and processing, and explored methods to account for GHG emissions or C-sequestration from LULUC. Our objectives were: to give an overview of current methods that are used in CFP assessment of feed ingredients; to demonstrate consequences of methodological choices on final CFPs of feed ingredients; to demonstrate sensitivity of CFPs of feed ingredients to technical in- and output data by performing a data sensitivity analysis.

2 Methods

2.1 Analysis framework

Six feed ingredients were used to demonstrate consequences of methodological choices and data sensitivity on CFPs of feed ingredients, i.e. wheat, maize, soybean meal, palm kernel expeller, rapeseed meal and beet pulp. We selected these ingredients because they are important ingredients in livestock concentrates, with major differences in nutritional value (Product Board Animal Feed 2008), and different production processes.

To assess the CFP of feed ingredients, the following activities along the production chain are of importance: production of the system inputs (e.g. fertilizers, pesticides, energy resources); cultivation and harvesting of crop products; drying and processing of crop products into single feed ingredients (this also includes the production of energy sources and auxiliary materials); processing of feed ingredients into a compound feed; transport of unprocessed and processed products between all activities, up to the farm were the feed is used for livestock. This study included all activities up to the gate of the factory responsible for drying and processing of the single feed ingredients.

Main greenhouse gases (GHGs) emitted during production of feed ingredients are CO_2, N_2O and to a lesser degree CH_4 (Duxbury 1994). Production of system inputs, such as synthetic fertilizers and energy resources, contributes mainly to CO_2 emission, whereas N_2O emissions are most important in crop cultivation. Emission of CH_4 is minor and mainly related to peat soils (IPCC 2006). Emissions from LULUC are dominated by CO_2. The CFPs of the six ingredients were computed by summing up emissions of these three gases based on their equivalence factor in terms of CO_2-equivalents (100-year time horizon): 1 for CO_2, 298 for N_2O, and 25 for CH_4 (IPCC 2007).

Methods and inventory data for calculating CFPs of ingredients were collected from literature and by contacting research institutes in France (INRA), Sweden (SIK) and the Netherlands (WUR). For wheat and rapeseed meal, data from several countries were used, resulting in nine data sets. Technical in- and output data for cultivation, drying, and processing of feed crops are included in Appendix 2.a. Yield and allocation factors of feed ingredients per feed crop are in Appendix 2.b.

2.2 Reference CFP

For each feed ingredient, a CFP in its most basic form was calculated, serving as a reference value to evaluate consequences of methodological choices and data sensitivity. Computations of the reference CFPs were based on the following assumptions. Emissions related to production of system inputs were based on life cycle inventories of the Ecoinvent database (2007). Production of seeds for sowing was not included. The amount of N from crop residues was based on IPCC (2006). Emissions of N_2O from crop cultivation were based on IPCC Tier 1 (IPCC 2006), which uses little or no country specific data. Emissions related to drying and processing of ingredients were based on Ecoinvent (2007). For transport, an average distance per ingredient was used, based on country of origin (Appendix 2.a), whereas transport emission factors (EFs) were taken from Ecoinvent (2007). Emissions related to LULUC were not included in the reference CFPs. They were treated as a methodological choice. In case of a multiple output system, we used economic allocation. Economic allocation implies that the impact of a certain process is allocated to the various products based on their relative economic value. This type of allocation is mostly used in CFPs of feed products. Allocation factors are in Appendix 2.b.

We demonstrated consequences of methodological choices and data sensitivity on CFPs of feed ingredients by comparing the CFP resulting from a single change with the reference CFP. Four categories of methodological choices were distinguished and are described in the following paragraphs: choices related to computation of emissions from cultivation (excl. LULUC), to emissions from LU, to emissions from LUC, and to emissions from processing. For emissions from cultivation and processing, we solely focused on emission calculations and not on the effect of changing the system boundaries, or allocation procedure. Although these aspects can have a large impact on the results, they have been subject to several other studies already (Flysjö et al., 2011b; Zehetmeier et al., 2012). For the data sensitivity analysis, the effect of a 10% change in various inventory data on CFPs was examined, while keeping the other parameters constant. Data used for the data sensitivity analysis are in Table 1. The meaning and relevance of the data are described in the method sections below.

2.3 Methods to compute GHG emissions from cultivation (excl. LULUC)

To calculate GHG emissions from cultivation (excl. LULUC), we need methods to determine N_2O emissions from cultivation, and to determine CO_2 emission from liming and urea fertilization. Emissions of N_2O from crop cultivation occur via a direct, and an indirect pathway. Direct N_2O emission follows from microbial nitrification and denitrification of N in the soil. Indirect N_2O emissions involve N that is removed from soils via volatilization (e.g. ammonia (NH_3) or nitrogen oxide (NO_x)), leaching or runoff (e.g. nitrate (NO_3^-) (IPCC 2006). CO_2 from liming and urea fertilization occurs via dissolving of carbonates (CO_3) in CO_2 and water (H_2O) (IPCC 2006). The

Table 1. Data used for data sensitivity analysis per category.

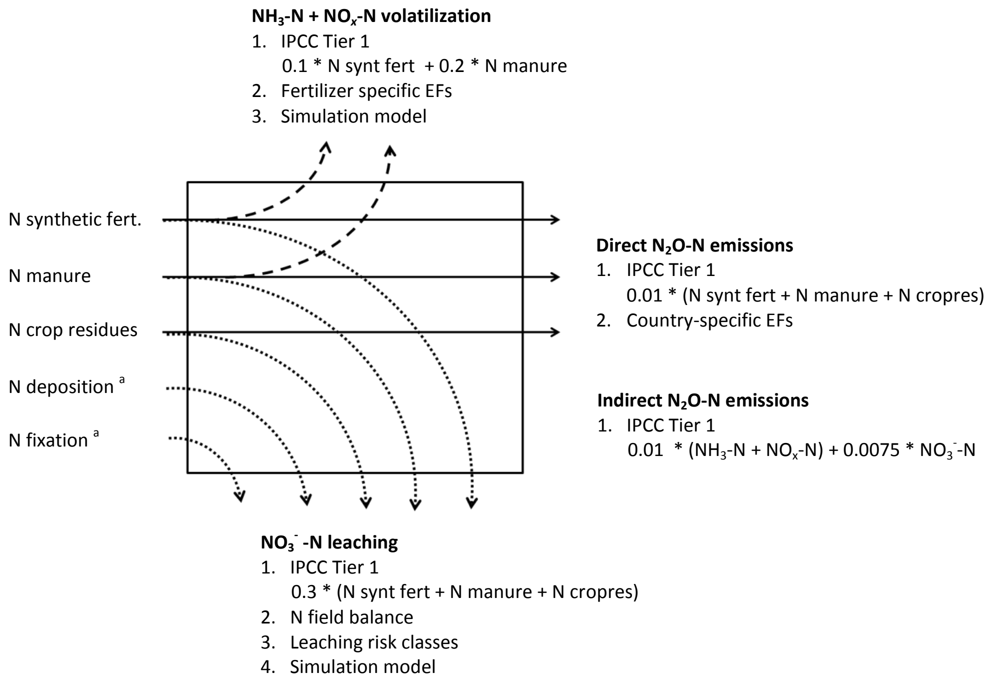
Cultivating and processing		Land use	Land use change	
Cultivating	Processing		Emissions per ha	Emissions per feed ingredient
Crop yield	Transport feed crops	Ref. soil C stock	C stocks before LUC[a]	Total C stock change
Synt. fertilizer N	Energy use drying	Soil C stock change factor	C stocks after LUC[a]	Area of LUC (ha)
Manure N	Energy use processing		Soil N stock before LUC	Amortization period
Crop residues	Product yield	Amortization period	Soil N stock after LUC	Allocation factor[b]
$CaCO_3$ (liming)	Price (allocation factor)		Amount of biomass burnt	
Diesel use				
Emission factors				

[a] Including C stocks in above and below ground biomass, dead organic matter, and the soil.
[b] Allocation of LUC emissions to logging.

literature review has revealed that such emissions are generally based on IPCC Tier 1, using an emission factor (EF) of 0.12 for limestone ($CaCO_3$), 0.13 for dolomite ($CaMg(CO_3)_2$), and 0.2 for urea, all expressed as kg CO_2-C per kg of product. No other methods for CO_2 emissions, therefore, were examined.

N_2O emissions

All peer-reviewed studies that calculated CFPs of feed ingredients that were found used IPCC (2006) to compute direct and indirect N_2O emissions. Some based their computations on general EFs as described in IPCC Tier 1, whereas others used country or fertilizer specific EFs, or simulation models (Tier 2 and 3). These methods are presented in Figure 1. Direct N_2O emissions depend on the amount of inorganic nitrogen (N) available in the soil. In crop cultivation the available inorganic soil-N increases due to the application of N fertilizers, the decomposition of crop residues, and the mineralization of soil-N through LULUC. The latter is considered in the



[a] Used for NO_3^- leaching-methods other than IPCC Tier 1.

Figure 1. Methods for direct and indirect N_2O emissions, and for NH_3+NO_x volatilization and NO_3^- leaching (i.e. used to calculate indirect N_2O emissions) in crop cultivation, and IPCC Tier 1 emission factors.

sections that describe the consequences of LULUC (§2.4 and §2.5). Only for the Netherlands, national inventory reports provide country specific EFs to calculate direct N_2O emissions (in kg N_2O-N per kg N applied; Van der Hoek et al. 2007), i.e. 0.005 for synthetic ammonium fertilizers, 0.02 for manure (incorporating into the soil), and 0.01 for crop residue-N (Van der Hoek et al. 2007). We, therefore, evaluated consequences of using these specific EFs on CFPs of Dutch feed ingredients only. Indirect N_2O emission is a function of volatilization of NH_3 and NO_x, and leaching of NO_3^- (Figure 1). To compute volatilization of NH_3 and NO_x two other methods and for leaching of NO_3^- three other methods were used besides Tier 1 (Figure 1).

NH_3 and NO_x volatilization

We compared two methods to compute NH_3 and NO_x volatilization with the reference situation. The reference situation uses one EF for synthetic fertilizer-N, and one for manure N (IPCC Tier 1). The two other methods are; (1) using fixed fertilizer specific EFs; and (2) using a simulation model (Figure 1). For fertilizer specific EFs for NH_3 volatilization per feed crop see Appendix 2.c. The simulation model resulted in a country specific, detailed prediction of NH_3 volatilization, taking into account the type of fertilizer, soil conditions, application technique, and seasonal influences such as weather conditions (Karlsson and Rodhe 2002). This method was available for feed ingredients from Sweden only. In IPCC Tier 1, NO_x volatilization is included in the EFs that are used, for the other two methods, NO_x volatilization was based on Ecoinvent (2007), i.e. 0.21 multiplied by direct N_2O-N emissions.

NO_3^- leaching

We compared three methods to compute NO_3^- leaching with the reference situation. The reference situation quantifies NO_3^- leaching as a fixed fraction of applied fertilizer-N and crop residue-N (IPCC Tier 1). The three other methods are: (1) the N-field balance; (2) the NO_3^- leaching risk classes method; and (3) using a simulation model (Figure 1). The N-field balance computes the difference between N-inputs and N-outputs at field level. This difference, also referred to as N surplus, is assumed to leach as NO_3^-, although in practice, several other factors influence NO_3^- leaching. The NO_3^- leaching risk classes method assumes optimal fertilization and determines NO_3^- leaching by assigning crops to one of four leaching risk classes, based on type of crop, succeeding crop, duration of period without a crop, and post-harvest soil-N content (Basset-Mens et al. 2007). Post-harvested soil-N content is based on literature and expert's opinion. Quantities of NO_3^- leaching per risk class are based on country specific models. For France, for example, risk classes include 15, 40, 70, or 100 kg NO_3^--N/ha (Basset-Mens et al. 2007). This method was only available for feed ingredients from France and for soybean meal. The simulation model results in a country specific, detailed prediction of NO_3^- leaching, taking into account the type of fertilizer, soil conditions, application techniques, and ground water level (SEPA 2008). This method was only available for feed ingredients from Sweden.

2.4 Methods to compute GHG emissions from land use

In our study, LU refers to changes in management of croplands. LU can contribute to GHG emissions by affecting soil-C stocks. An increase in soil C indicates removal of CO_2 from the atmosphere (C-sequestration), whereas a decrease indicates CO_2 emission. In addition, a decrease in soil C leads to N mineralization and hence N_2O emission (IPCC 2006; Vellinga et al. 2004). It is assumed that when land use type and management system remain unchanged for decades, the soil C stock will no longer increase of decrease and stabilization is reached. In our reference situation, therefore, no changes in soil C, and hence, no LU emissions were assumed.

Parameters that affect soil C, and hence cause emissions from LU, are changes in the level of C inputs and changes in management practices that disturb the soil structure, such as tillage (Ogle et al. 2012). The level of C-input highly depends on the amount of crop residues remaining on the field, depending on crop yield and crop residue removal. Manure application also is a source of C input. Changes in management practices that affect crop yield, such as a change in irrigation or fertilization regime, are related to LU emissions due to their indirect effect on the amount of crop residues (IPCC 2006). Changing to no tillage has been suggested as a strategy to decrease decomposition rates and increase C-sequestration (Zotarelli et al. 2012). Recently, however, this effect has been questioned as a change in tillage system also can affect crop yield in a positive or negative direction, and hence C-input as well (Ogle et al. 2012).

Literature review shows that, so far, LU emissions have not been included in CFPs of feed ingredients. Assessing LU emissions requires detailed information on current and historical management practices, which is often not available. Furthermore, methods to calculate soil-C stock changes have high levels of uncertainty. To gain insight into the potential consequences of including LU emissions on CFPs of feed ingredients, we calculated the effect of a change in tillage system on soil-C stock levels based on IPCC Tier 1 methods and default values (IPCC 2006). To estimate the effect of a change in a tillage system, we need a reference tillage system. This was no tillage for soybeans, and full tillage for all other feed crops. A change in tillage system, therefore, means changing to reduced or full tillage for soybeans, and changing to reduced or no tillage for all other feed crops. Palm kernel expeller was excluded from the analysis as palm fruit is a perennial crop and does not require tillage.

Emissions from LU depend on the level of soil C in the reference situation. For soybeans the reference soil C stock in the top layer (0-30 cm) was assumed to be 35 t C/ha, for all other crop it was 32 t C/ha, based on estimates for C stocks in cropland after LUC from native vegetation with an average of 60 t C/ha (IPCC 2006). C stocks in soybeans were higher because as a default, no-tillage is assumed to result in higher soil C stocks than full tillage (IPCC 2006). The C stocks were based on very rough estimates, but in line with 36 t C/ha in cropland in the Netherlands from Vellinga and Hoving (2011), and 28 t C/ha in cropland in Brazil from Cederberg et al. (2011). Emissions were amortized over a period of 20 years (IPCC 2006).

2.5 Methods to compute GHG emissions from land use change

In our study, LUC refers to transformation of non-cropland, such as forest land, scrubland, and natural grassland, into cropland. LUC can contribute to GHG emissions by affecting C-stocks in the ecosystem, including C stocks in above and below ground biomass, dead organic matter, and soil organic matter. An increase in C stocks contributes to C sequestration, whereas a decrease contributes to CO_2 emissions. CO_2 emissions, for example, can occur from (incomplete) burning of above ground biomass (e.g. deforestation), from decay of biomass, and from changes in soil-C. In addition, changes in soil-C can lead to N_2O emissions (Vellinga et al. 2004), and burning of biomass leads to N_2O and CH_4 emissions.

To calculate LUC emissions related to crop cultivation, we need methods to estimate GHG emissions per ha LUC (i.e. amount of emissions resulting from transforming one ha of non-cropland into cropland), and we need to allocate LUC to a specific crop (i.e. how many hectares are changed, which part of LUCs are allocated to which crop).

Estimating GHG emissions per ha of LUC

Methods that estimate GHG emissions per ha of LUC generally quantify changes in C-stocks (Searchinger et al. 2008; Leip et al. 2010; Cederberg et al. 2011). They vary in type of emissions accounted for, and in time period over which changes in C-stocks are examined (Appendix 2.d). We studied the consequences of including or omitting different types of emissions by evaluating the contribution of each type of emission to total LUC emissions per ha, for situations relevant to feed crops. These were: changing tropical forest, scrubland, and natural grassland into annual cropland in Brazil (i.e. relevant for soybean meal), and changing tropical forest into perennial cropland in Malaysia (i.e. relevant for palm kernel expeller). Calculations were based on IPCC Tier 1 methods and default values. For C-stocks in different land use categories see Appendix 2.e. We assumed that part of the above ground biomass was burned, i.e. 36% of 160 t DM/ha biomass in tropical forest, 72% of 14.3 t DM/ha biomass in scrubland, and 92% of 5.2 t DM/ha biomass in grassland (IPCC 2006).

We studied the consequences of a difference in time period over which changes in C stocks are examined by comparing the annual balance method (IPCC 2006), with the net committed emissions method (Fearnside 1997; Cederberg et al. 2011). The annual balance method is used most commonly and focuses on a specific time period, i.e. the moment that the land is cleared and used for another purpose, e.g. cropland. It does not include delayed emissions or C-sequestration other than in the first year after LUC. The net committed emissions method encloses a longer time period, and includes all delayed emissions and C-sequestration that take place after the initial LUC. For soybean production in Brazil, for example, this method accounts for the fact that part of the land that was initially cleared for soybean (or pasture) production is abandoned after a few years. This abandoned land may regenerate into secondary forest, which can sequester C in biomass and soil, but it also means that more than one ha of land is changed to provide one new ha of soybeans in permanent production. Differences between the two methods were analyzed for deforestation of tropical forest for soybean production in Brazil. Land use dynamics for the net committed emissions method were based on Macedo et al. (2012), assuming that 15% of the deforested land was abandoned in a later stage. As 34% of the deforested land could not be classified into a land use category (Macedo et al. 2012), this 15% was increased up to 23%, assuming that similar transition probabilities hold for the unclassified category. We assumed that abandoned land regenerates into secondary forest, and that the proportion of land that was initially deforested for crop production and later transformed into pasture was negligible. Emissions per ha of LUC were calculated similarly according to the annual balance method (IPCC 2006).

Allocation of LUC to a specific crop

To allocate LUC to a specific crop, we need to decide which crops are responsible for which part of the LUCs. Methods that are described in literature show high variation. We compared three methods. **Method 1** focuses on direct LUC within a country or region, and allocates emissions to the crops that are directly related to the LUC (Jungbluth et al. 2007; Prudencio da Silva et al. 2010). Soybean meal and palm kernel expeller were the only two feed ingredients related to direct

LUC. For soybean meal, we assumed that 1% of the soy produced in Central West Brazil comes from tropical forest, and 3.4% comes from scrubland, whereas soy from South Brazil does not contribute to LUC (Prudencio da Silva et al. 2010). For palm kernel expeller, we assumed that 100% of the palm area in Malaysia comes from tropical forest (Jungbluth et al. 2007). Calculation of emissions per ha of LUC are described in the former section. For soybean meal, emissions per ha were based on the annual balance method and the net committed emissions method, whereas for palm kernel expeller, the annual balance method was used only. Amortization period was 20 years.

As opposite to method 1, the following two methods also included indirect LUCs. **Method 2** was based on Leip et al. (2010), and focuses on LUCs within a country, or country block (i.e. a group of countries), after which emissions were averaged for European Union (EU) countries and non-EU countries. For each country (block) the total area of LUC was determined for a specific time period, and emissions were allocated to the crops that showed an increase in total cropland in that time period, based on their relative contribution. Different types of LUCs were included, such as the transformation of natural grassland, scrubland, and tropical forest into cropland. LUC emissions were averaged, resulting in one weighted value per crop (product) from the EU, and one weighted value per crop (product) from non-EU countries. Emission calculations were based on the annual balance method, but did cover the total area of LUC (i.e. all LUC was included). They were mainly based on IPCC default values (IPCC 2006), and for tropical forest and scrubland similar to method 1. Amortization period was 20 years.

Method 3 was based on Audsley et al. (2009). This method considers total LUC emissions worldwide, and allocates it to all agricultural land in use for commercial food production. Emissions were derived from Barker et al. (2007), and included GHG emissions and C-sequestration from forestry only. C-sequestration was included as the method accounts for afforestation too. No amortization was applied. The method resulted in a single emission factor of 1.43 t CO_2/ha of agricultural land.

2.6 Processing of feed ingredients

Feed ingredients can originate from crops directly (e.g. wheat and maize), or from industrial processing of crops (e.g. palm kernel expeller, soybean meal, rapeseed meal and beet pulp). Ingredients that derive from industrial processing are often by-products from the biofuel or food industry. Rape seed meal, for example, is a by-product from the processing of rape seeds, whereas beet pulp is a by-product from the processing of sugar beets. To compute the CFP of a feed ingredient, therefore, methods that deal with multiple output systems are required. We used economic allocation in case of a multiple output system (see section 2.2). The impact of the processing stage, therefore, is determined by the amount and type of energy and auxiliary materials, and the emissions factors that are used. For allocation, product yield after processing and price data are important. Overall, the main input for processing is energy (electricity, natural gas, diesel), facilitating processing stages as washing (sugar beets), crushing (oil seeds), and drying (grains, meal and pulp). Hexane is often used in the oil industry as a solvent extraction, but generally the use of auxiliary materials is limited. Division of the processing stage into sub-processes increases the accuracy of CFP studies, but is often limited by lack of data (ISO 14043, 2000). By changing the allocation factor of processing in the sensitivity analysis (see Table 1), we show for which feed ingredients sub division is most important. A more detailed description of processing of feed ingredients can be found in Jungbluth et al. (2007).

3 Results and discussion

3.1 Reference carbon footprint

Figure 2 shows the reference CFP of feed ingredients and the fractional contribution of different processes. For main products, such as maize and wheat, and for unprocessed products, N_2O emissions from cultivation and production of synthetic fertilizer-N are by far most important contributors (>65%). For feed crops that use little synthetic fertilizer-N (e.g. legumes such as soybeans), or use different management practices that dominate emissions (e.g. high levels of irrigation such as palm fruit) this differs. Besides N_2O emissions and synthetic fertilizer-N, production and combustion of diesel is quite important (10%). Other aspects in cultivation (production of P_2O_5 and K_2O fertilizers, pesticides and machinery) have a minor contribution only (<5%). Emissions from drying and transport (about 10% of the CFPs) increase as the difference between DM content of harvested and dried product increases, or when transport distances increase.

For by-products, such as palm kernel expeller and soybean meal, processing stages are important. Also for by-products, however, N_2O emissions from crop cultivation and production of synthetic fertilizer-N are important contributors, except when the (economic) allocation factor for assigning emissions from crop cultivation to the feed ingredient is low and further processing is required (e.g. beet pulp).

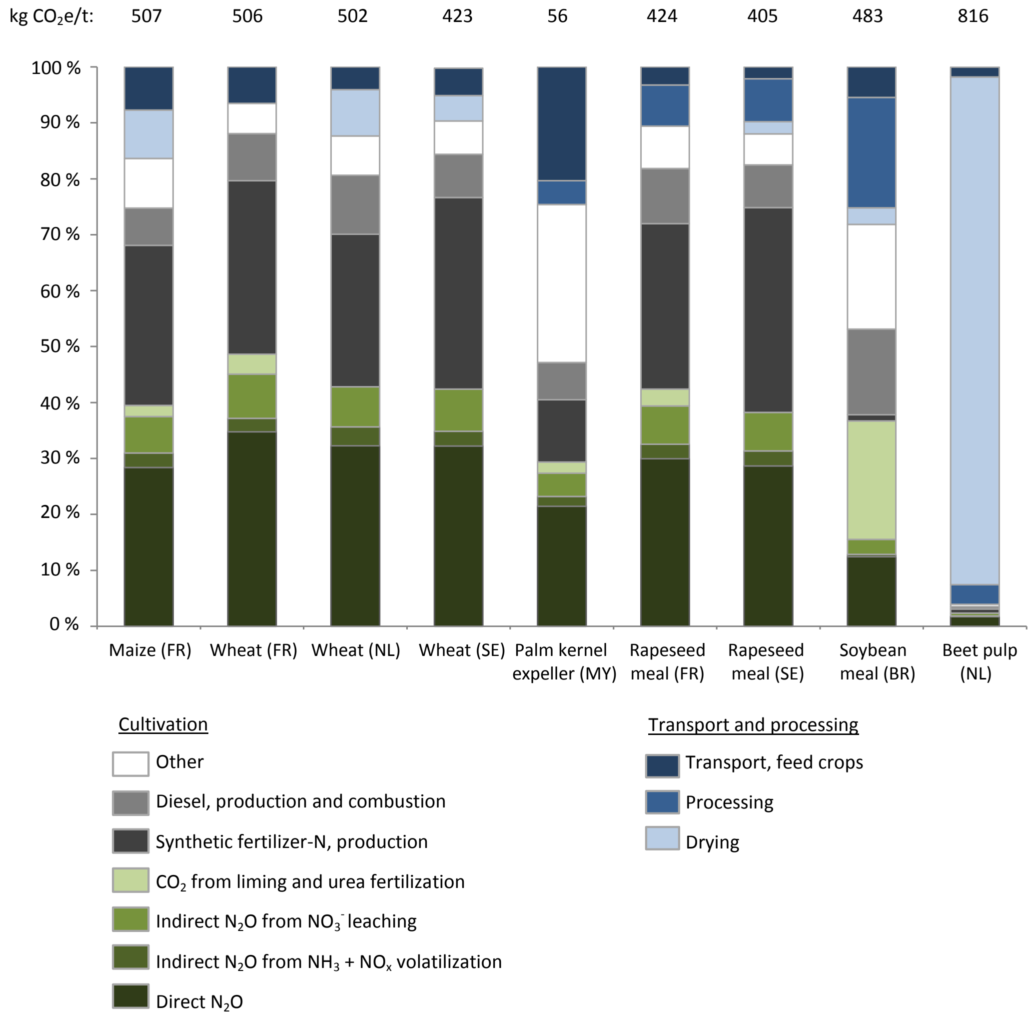


Figure 2. Reference CFP of feed ingredients and fractional contribution of different processes.

Only activities up to drying and processing of the single ingredients were included. For several ingredients, transport might have had a bigger impact, and CFP might have been higher, when all activities up to the country of final destination would have been included. This counts especially for feed ingredients from tropical areas that are exported to Europe, such as soybean meal and palm kernel expeller. In Prudencio da Silva et al. (2010), for example, shipping of soybeans from South Brazil to the Netherlands contributed 23% to the total CFP of these soybeans.

3.2 Cultivation and processing (excl. LULUC)

Methods for N_2O emissions from crop cultivation

Table 2 shows the reference CFP of feed ingredients and consequences of using more specific methods to calculate N_2O emissions from cultivation. Country specific EFs for direct N_2O emissions were available for two feed ingredients only. Using this method changed CFPs with only 0 and 2%. For wheat (NL), a decrease in emissions resulting from a lower EF for synthetic fertilizer N compared to the reference, was leveled out by an increase in emissions resulting from a higher EF for manure (Van der Hoek et al. 2007). For feed ingredients that use another ratio of synthetic and organic fertilizer, the relative change could increase.

Indirect N_2O emissions were computed as 0.1 x (NH_3 + NO_x) + 0.0075 x NO_3^-. This means that CFPs will be changed only when a change in method substantially alters the amount of NH_3 + NO_x, or NO_3^-. Using more specific methods to estimate volatilization of NH_3 + NO_x changed CFPs only with 2%, whereas using more specific methods to quantify NO_3^- leaching changed CFPs with -7 to +12%. Based on these results, correct assessment of NO_3^- leaching is most important when calculating N_2O emissions from crop cultivation.

Table 2. Reference CFP[a] of feed ingredients (kg CO_2e/t) and consequences of using more specific methods to calculate N_2O emissions from cultivation (%).

		Direct N_2O emission	NH_3 + NO_x volatilization[b]		NO_3^- leaching[b]		
Feed ingredient	kg CO_2e/t	Country spec. EFs (%)	Fertilizer spec. EFs (%)	Simulation model (%)	N-field balance (%)	Leaching risk class (%)	Simulation model (%)
Maize (FR)	507		-2		-5	+6	
Wheat (FR)	506		-1		-7	-4	
Wheat (NL)	502	+2	-2		-3		
Wheat (SE)	423		-2	-2	-3		-4
Palm kern. exp. (MY)	56		0		+9		
Rapeseed meal (FR)	424		-1		-2	-3	
Rapeseed meal (SE)	405		-1	-2	-2		-3
Soybean meal (BR)	483		0		+1	+12	
Beet pulp (NL)	816	0	0		0		

[a] The reference CFPs are based on IPCC Tier 1 methods.
[b] Contributing to indirect N_2O emissions.
No value means no data available.
EFs = emission factors

Data sensitivity of inventory data

Table 3 shows the reference CFP of feed ingredients and consequences of a 10% change in inventory data and EFs. Overall, changing crop yield and synthetic fertilizer-N changed CFPs most. Results correspond exactly with results in Figure 2. This means, the higher the contribution of a certain aspect to the CFP of an ingredient, the higher the impact of the relative change. Crop yield is related to the contribution of cultivation in total. Therefore, CFPs of main products (maize and wheat), and (by-) products that have little emissions from processing (rapeseed meal), are more sensitive to a change in crop yield than products in which processing is more important (palm kernel expeller, soybean meal, and beet pulp). Similarly, we can explain the relative change of CFPs due to a 10% change in amount of synthetic fertilizer-N, affecting emissions from production and application, i.e. important contributors for most ingredients (Figure 2); and the relative change of the CFP of beet pulp due to a change in energy use for drying. Consequences of a change in product yield after processing are higher when the contribution of processes after processing are minor and when the product has a high allocation factor (e.g. soybean meal). Consequences of a 10% change in price were highest for feed ingredients with a low allocation

Table 3. Reference CFP of feed ingredients (kg CO_2e/t) and consequences of a 10% change in inventory data and emission factors (EFs) (%).

		Cultivation										Transport & Processing				
Feed ingredient	kg CO_2e/t	Crop yield (%)	Synthetic fert. N (%)	Manure N (%)	Crop residues N (%)	$CaCO_3$ (liming) (%)	Diesel (%)	EF direct N_2O (%)	EF indirect N_2O (%)	EF NH_3+NO_x volat. (%)	EF NO_3^- leaching (%)	Transport, feed crops (%)	Energy drying (%)	Energy process. (%)	Product yield (%)	Price (%)
Maize (FR)	507	8	6	1	1	0	1	3	1	0	1	1	1	-	-	0
Wheat (FR)	506	9	6	0	1	0	1	3	1	0	1	0	-	-	-	0
Wheat (NL)	502	8	5	1	1	-	1	3	1	0	1	0	1	-	-	1
Wheat (SE)	423	9	6	1	1	-	1	3	1	0	1	0	0	-	-	1
Palm kern. exp. (MY)	56	7	3	0	0	-	2	2	0	0	0	2	-	0	0	10
Rapeseed meal (FR)	424	8	6	0	1	0	1	3	1	0	1	0	-	1	2	7
Rapeseed meal (SE)	405	8	6	1	1	-	1	3	1	0	0	0	0	1	2	7
Soybean meal (BR)	483	7	1	0	2	2	2	1	0	0	0	0	0	2	6	4
Beet pulp (NL)	816	0	0	0	0	-	0	0	0	0	0	0	9	0	0	1

- = not applicable

factor. Thus, for these ingredient, division of the processing stage into sub-processes can have a major impact on CFPs. A 10% change in other technical in- and output data hardly affected CFPs. Regarding the EFs, results show that a 10% change in the EF for direct N_2O emissions changed CFPs most (about 3%).

In cultivation, the quantitative order, thus relative importance of inputs per crop type does not vary between studies. For most feed ingredients, therefore, high-resolution data for crop yield and synthetic fertilizer-N are most important for correct CFP assessment. For imported feed ingredients, means and distance of transport can be paramount. In general, higher accuracy in CFPs can be achieved by analyzing the relative contribution of different processes, and validating data for those processes that have a major contribution.

3.3 Land use

Methods for emissions from LU

Table 4 shows the reference CFP of feed ingredients and consequences of a change in tillage system. To evaluate LU methods, the default scenarios are used. Changing from full to reduced tillage changed CFPs by -1 to -15%; changing from full to no tillage by -1 to -28% (Table 4). Changes in CFPs were lowest for feed ingredients from France, because EFs varied with moisture regime and were relatively low for France, which has a dryer climate than the Netherlands and Sweden (IPCC 2006). Changing to reduced tillage resulted in C-sequestration of about 90 kg C/ha/year for France, and 140 kg C/ha/year for the Netherlands and Sweden. When changing to no tillage this was 220 kg C/ha/year for France, and 260 kg C/ha/year for the Netherlands and Sweden. These numbers are in line with results found by Ogle et al. (2012). For soybean meal (using no tillage in the reference situation), changes were more pronounced, i.e. +55% for changing to reduced tillage and +81% for changing to full tillage, because EFs for tropical and wet climates were higher compared to European climates, and soybeans have a relative low yield compared to other feed crops. For soybeans, changing to reduced tillage resulted in a soil-C loss of about 260 kg C/ha/year, whereas for changing to full tillage this was 385 kg C/ha/year. This is in line with results found by Zotarelli et al. (2012).

Effects of a change in tillage system on soil-C stocks have been questioned (Ogle et al. 2012). Ogle et al. (2012) showed that the final effect of a change in tillage system depends on a combination of crop type, climate, soil type, fertilization level and other aspects, and can vary between years due

Table 4. Reference CFP of feed ingredients (kg CO_2e/t) and consequences of a change in tillage system (%) (default soil-C stock change factor (-10% ; +10%)).

	Ref. CFP (kg CO_2e/t)	Consequences of a change in tillage system (%)			
Feed ingredient	Full tillage	Reduced tillage		No-tillage	
Maize (FR)	507	-8	(+7 ; -8)	-19	(-17 ; -20)
Wheat (FR)	506	-8	(+9 ; -9)	-21	(-19 ; -23)
Wheat (NL)	502	-11	(+4 ; -12)	-20	(-18 ; -23)
Wheat (SE)	423	-15	(+5 ; -16)	-28	(-25 ; -30)
Rapeseed meal (FR)	424	-9	(+10 ; -9)	-21	(-19 ; -23)
Rapeseed meal (SE)	405	-12	(+4 ; -13)	-23	(-20 ; -25)
Beet pulp (NL)	816	-1	(0 ; -1)	-1	(-1 ; -1)
	No tillage	Reduced tillage		Full tillage	
Soybean meal (BR)	483	+55	(+49 ; +61)	+81	(+73 ; +89)

to variation in e.g. weather conditions. In cold and wet climates, changing from full to no tillage can even result in a decrease in soil-C stock levels (Ogle et al. 2012). IPCC (2006) provides an uncertainty range along with their default values that displays this variation. Results shown here, therefore, are a first rough estimate of possible changes in CFPs when including LU emissions, but do not cover the complexity that is required for a detailed evaluation. For a detailed evaluation, all different aspects that influence crop yield, C-input, and soil-C stock levels need to be included and assessed. A change in crop yield is of particularly interest as this will also affect the allocation of emissions from cultivation.

LU emissions due to a change in tillage system are non-recurrent, whereas N_2O emissions from N application are annual. LU emissions were amortized over a period of 20 years (IPCC 2006), thus after 20 years CFPs are no longer affected. This 20 years period is arbitrary and an estimation of the time that it takes to get to a new soil-C balance. Including consequences of a change in tillage system means that this change is assumed to be permanent. If not, CO_2 that is sequestered from the atmosphere is emitted again as soon as the 'old' tillage system is re-implemented.

Data sensitivity in LU emissions

To evaluate data sensitivity, results from a 10% change in the default soil-C stock change factor are compared. These results are presented between brackets (Table 4). In the default scenario, the soil-C stock is multiplied by the default stock change factor of 1.05 when changing from full to reduced tillage, in case of feed crops from France (IPCC, 2006). For feed crops from the

Netherlands and Sweden this factor is 1.08. A change of -10% means that these stock change factors become <1, i.e. the soil-C stock decreases, resulting in CO_2 emissions instead of C-sequestration. A -10% change in the default stock change factor, therefore, increases CFPs, whereas a +10% change decreases CFPs (Table 4). As the stock change factor is subject to a lot of uncertainty (IPCC 2006), CFPs that include emissions from a change in tillage system should be interpreted carefully. For other feed ingredients, the default stock change factor was considerably higher than 1, and a 10% change did not change results from C uptake into C losses, or vice versa. In such cases, results from a 10% change in stock change factor also apply to 10% change in soil-C stock level, whereas a 10% change in amortization period resulted in slightly lower changes. This can be explained by the function (C stock change factor x C stock level / amortization period). Consequences increased with an increase in relative impact of LU emissions in CFPs (i.e. results at the default stock change factor) and, therefore, were highest for soybean meal.

3.4 Land use change

Methods for emissions from LUC

Table 5 shows the contribution of different emissions to total LUC emissions per ha for different land use transitions. Emissions from LUC are dominated by CO_2 emissions. When changing tropical forest or scrubland into cropland, the majority of the CO_2 emissions result from changes in above ground-C, below ground-C and soil-C. When changing grassland into cropland, the majority of the CO_2 emissions result from changes in soil-C, but CO_2 emissions from changes in above and below ground-C, and N_2O from changes in soil-N were still quite important. Excluding one of these emissions would result in underestimation of LUC emissions. When including emissions from burning, part of the C in biomass will be emitted as CO and CH_4, and can therefore no longer be emitted as CO_2. The net contribution of N_2O and CH_4 emissions from burning of biomass (i.e. after correction for foregone CO_2 emissions) was minor (Table 5). Without this correction, emissions from deforestation increased with about 10 t CO_2e/ha, i.e. 1% of total LUC emissions/ha.

There is little information on C-stock levels in soils and below ground biomass in perennial croplands. The default soil-C stock change factor for transformation of natural land into perennial cropland is 1, which means no change in the long term (IPCC Tier 1). This default value has a high uncertainty (50%), and because in this case soil-C losses are more likely than C-sequestration, emissions from changes in soil-C might be underestimated. In addition, CO_2 emissions from

Table 5. Total LUC emissions per ha (t CO_2e/ha) and contribution of different emissions (%) for different land use transitions.

Country	Initial land use	Final land use	Total t CO_2e/ha	CO_2 from above ground C (%)	N_2O+CH_4 from burning biomass[a] (%)	CO_2 from below ground C (%)	CO_2 from dead organic matter C (%)	CO_2 from soil-C (%)	N_2O from soil-N (%)
Brazil	tropical forest	annual cropland	825	63	0	23	2	11	1
	scrub land	annual cropland	297	47	0	19	0	31	3
	natural grassland	annual cropland	128	8	0	13	0	71	7
Malaysia	tropical forest	perennial cropland	496	97	1	NAV	3	0	0

[a] Corrected for foregone CO_2 emissions from biomass due to emission of CO and CH_4 from burning.
NAV = not available

changes in below ground biomass-C were not included, which is also expected to be an underestimation.

Frequently, LUC emissions are amortized over a period of 20 years. When applying amortization, it seems correct to include land use transitions and C-sequestration over the same period. For annual croplands, in which C-sequestration is negligible, this will only affect results when part of the cropland is changed into another land use type, or abandoned during the amortization period. This would mean that more than one ha of land is transformed to provide one ha of cropland. The net committed emissions method accounts for such land use transitions and delayed emissions and C-sequestration after the LUC. When applying amortization, therefore, this method seems to be most suitable. Lack of information on land use transitions, however, can hamper its use.

Using the net committed emissions method for changing tropical forest into annual cropland in Brazil, resulted in an emission of 778 t CO_2e/ha (775 t from a change in C stocks, and 3 t from burning including a correction for foregone CO_2 emissions), compared to 825 t CO_2e/ha for the annual balance method (Table 5). Per ha of permanent cropland, however, 1.23 ha is deforested. Hence, total LUC emission per ha of permanent cropland is 957 t CO_2e.

Table 6 shows the reference CFP of feed ingredients and consequences of including LUC emissions using three methods. For method 1, the difference between the annual balance method and the net committed emissions method are given also. Method 1 focuses on direct LUC. Including direct LUC increased the CFP of soybean meal with 35-38%, whereas the CFP of palm

Table 6. Reference CFP of feed ingredients (kg CO_2e/t) and consequences of including LUC emissions using three different methods, with for method 1 the difference between the annual balance method (AB) and the net committed emissions method (NCE) (%).

	CFP without LUC	Consequences of including LUC emissions (%)			
		Method 1		Method 2	Method 3
	kg CO_2e/t	AB	NCE		
Maize (FR)	507	-	-	+3	+31
Wheat (FR)	506	-	-	+8	+40
Wheat (NL)	502	-	-	+8	+31
Wheat (SE)	423	-	-	+10	+42
Palm kernel exp. (MY)	56	+877	NAV	NAV	+52
Rapeseed meal (FR)	424	-	-	+69	+42
Rapeseed meal (SE)	405	-	-	+73	+37
Soybean meal (BR)	483	+35	+38	+632	+82
Beet pulp (NL)	816	-	-	NAV	+2

Methods: 1= direct LUC (this study), 2= Leip et al., 2010, 3=Audsley et al., 2009.
NAV = not available

kernel expeller increased with 877%. Method 2 and 3 also include indirect LUC. Including indirect LUC via method 2 (Leip et al. 2010) mainly affected the CFP of those ingredients that expanded their cultivation area over the last ten years, i.e. rapeseed meal (change in CFP is about +70%) and soybean meal (change in CFP is +632%). Including indirect LUC increased especially the CFP of soybean meal, because method 2 includes LUCs related to the expansion of soybean cultivation in 24 non-EU countries (blocks). This value, therefore, included the significant increases in area of soybean cultivation in the whole of Brazil, but also outside Brazil, for example in Venezuela (Leip et al. 2010). Method 3 (Audsley et al. 2009) uses one single EF per ha of land. The lower the yield per ha and the higher the allocation factor, the higher the emissions per kg of feed ingredient. Including indirect LUC via method 3 changed CFPs of wheat and maize with about +40%. This is much more than with method 1 or 2, whereas for soybean meal and palm kernel expeller the change in CFPs was less than for method 1 and 2.

LUC emissions generally dominate CFPs, but the final change in CFPs varies between methods. There is no shared consensus, and the method chosen will greatly affect the outcome. The best method depends on the objective of the study. To encourage individual companies or countries to invest in sustainable production and to stimulate them to reduce deforestation, the method should focus on the direct link between products and LUC, i.e. method 1. Stimulating individual companies could lead to the combined demand of many actors for more sustainable production (i.e. no deforestation), and hence to reduced deforestation in the long term (Weidema, 2003). When the objective, however, is to emphasize that because of globalization of food and feed markets, the agricultural sector as a whole is responsible for deforestation, than the method should not differentiate between direct and indirect LUC. In this case, every ha of land used for commercial production purposes should be allocated a share of LUC emissions (method 3; Audsley et al. 2009). Method 3 will stimulate efficiency and increasing crop yield, and will favor feed crops from regions where the growth potential is highest due to optimal agro-ecological circumstances, because reducing land use requirements is the only option to reduce LUCs emissions, and hence CFPs. This method, however, does not provide a strong direct incentive to reduce deforestation.

As LUC emissions dominate CFPs of feed ingredients, including these emissions might diminish the incentive to reduce emissions from cultivation and production other than from LUC. To avoid this, and because of high uncertainty and variation in calculating LUC emissions (method are highly debated, and there is no shared consensus), emissions from LUC should be presented separately from other emissions in CFPs (Flysjö et al. 2012). Moreover, this seems correct because LUC emissions are non-recurrent and only affect CFP for a certain period (i.e. dependent on amortization period), whereas other emissions from cultivation and processing recur annually.

Data sensitivity in LUC emissions

Table 7 shows the total LUC emissions per ha for different land use transitions and consequences of a 10% change in input data needed to calculate these emissions. Except for the amount of biomass burned, all inputs are directly related to the level of C-stocks they refer to. This means that the higher the C-stock, the more paramount the consequences of a 10% change in this C-stock will be. When changing tropical forest into cropland, for example, above ground biomass in the initial land use was the largest C-stock (Appendix 2.e), and a 10% change in above ground biomass, therefore, affected emissions most (Table 7). Similarly, for scrubland this is the soil-C stock in the initial land use, and for grassland this is the soil-C stock in both the initial and final land use.

We examined the sensitivity of CFPs that include direct LUC emissions (i.e. method 1) to changes in input data necessary to calculate LUC emissions per feed ingredient. The more relevant the data for computing LUC emissions are (Table 5), and the more dominant LUC emissions are in CFPs (Table 6), the larger the effect of a change in data, with a maximum of 10% (i.e. equal to the change in data). A 10% change means that the aspect is highly relevant and that the CFP is

Table 7. Total LUC emissions for different land use transitions (t CO_2e/ha) and consequences of a 10% change in different input data (%).

			Total emissions	Above ground biomass C stocks		Biomass burnt	Below ground biomass C stocks	Dead organic matter C stocks	Soil C stocks		Soil N stocks	
Country	Initial land use	Final land use	t CO_2e/ha	Initial (%)	Final (%)	Initial (%)	Initial (%)	Initial (%)	Initial (%)	Final (%)	Initial (%)	Final (%)
Brazil	tropical forest	annual crops	825	6	NAP	0	2	0	3	1	0	0
	shrub land	annual crops	297	5	NAP	0	2	0	8	5	1	0
	natural grassland	annual crops	128	1	NAP	0	1	NAP	17	8	2	1
Malaysia	tropical forest	perennial crops	496	10	1	0	NAV	0	4	4	1	1

NAP = not applicable
NAV = not available

completely determined by LUC emissions. As LUC emissions are almost completely determined by a change in C stocks (Table 5), the consequence of a change in total C stock was comparable to the consequence of a change in the area of LUC (3% for soybean meal and 9% for palm kernel expeller in both cases). Consequences of a change in amortization period were slightly less, but about the same (3% for soybean meal and 8% for palm kernel expeller), which can be explained by the function (emissions per ha x area of LUC / amortization period). A 10% change in allocation factor means that 10% of the above ground biomass was allocated to logging, instead of no allocation. For perennial croplands, LUC emissions were for 97% determined by a change in above ground biomass-C, whereas for annual cropland this was 63% (Table 5). Therefore, a change in allocation factor will affect the CFP of a perennial crop more than the CFP of an annual crop. In this study, a change in allocation factor changed the CFP of soybean meal by 1%, whereas the CFP of palm kernel expeller was changed by 9%.

For accurate evaluation of LUC emissions, CO_2 from changes in above ground biomass, below ground biomass, and soil-C should be included. For correct interpretations and comparisons of LUC emissions per feed ingredient, it is equally important to consider assumptions on the area of LUC, as C-stock levels and amortization period. Assumptions about logging can be important too, especially for perennial croplands.

4 General discussion

We did a sensitivity analysis to identify for which data the outcome of the CFP studies are most sensitive. Such information can improve the accuracy of CFP studies as it points out which data should be considered first. The effect of a change in input data on the outcome of the study is determined by the magnitude of the change. Changing the input data with 50%, or 5%, instead of 10%, however, does not change the priority of the input data, and will therefore not affect the conclusions of the sensitivity analysis.

Results do not give insight into the effect of data uncertainty, which refers to uncertainty due to inaccurate measurements, or lack of data; or data variability, which refers to variation in the real world, e.g. temporal and spatial variation (Huijbregts, 1998). An uncertainty analysis requires information on distribution and data quality indicators, and can be performed with, for example, a Monte Carlo analysis (Heijungs and Huijbregts, 2004). The effect of data variability can be large, but is not a matter of lack of data quality or knowledge. Crop yield, for example, varies greatly between countries, but also within countries and between years (FAOSTAT, 2010), and has a large impact on the CFP of feed ingredients. The same accounts for application of manure and fertilizers, including limestone. The effect of data uncertainty and variability, therefore, can be much larger than results shown by our sensitivity analysis. It is important to realize this when comparing CFP studies. Particularly, emissions from LULUC can vary greatly due to high levels of data uncertainty.

5 Conclusions

We explored the consequences of methodological choices and data sensitivity on CFPs of feed ingredients, for emissions from cultivation and processing, land use (LU), and land use change (LUC). Calculation methods for direct and indirect N_2O emissions from cultivation were consistent among studies, whereas differences in methods to calculate NH_3 and NO_x volatilization (contributing to indirect N_2O emissions) hardly affected CFPs. Differences in methods to calculate NO_3^- leaching (also contributing to indirect N_2O emissions), however, can affect CFPs considerably. High-resolution data was most important for crop yields and the quantity as well as the type of synthetic nitrogen fertilizer. For by-products, data on processing and transport can be paramount. Higher accuracy in CFPs can be achieved by analyzing the relative contribution of different processes and validating data for the most important parameters, e.g. yield and N-fertilizer data.

We explored the consequences of including LU emissions (i.e. emissions due to a change in management practices) by assessing the effect of a change in tillage system. Results show that changing to no-tillage can potentially reduce CFPs. For a detailed evaluation, however, all aspects that affect crop yield, C-input and soil-C stock levels should be included. For accurate evaluation of LUC emissions, CO_2 from changes in above ground biomass, below ground biomass, and soil-C should be included. The net committed emissions method seems to be most appropriate when applying amortization: C-stock changes and land use transitions are accounted for preferably over the same period as the amortization period. For allocating LUC to different crops, the objective of the study is important and the method will greatly affect results. LULUC emissions should be presented separately from other emissions, because there is no consensus about the method to calculate these emissions, and LULUC emissions are non-recurrent, whereas other emissions reoccur annually. To compare LUC emissions per feed ingredient the area of LUC, C-stock levels, and amortization period should be considered. Assumptions about logging can be important too, especially for perennial croplands.

Variability in methods and data can considerably affect CFPs of feed ingredients, and hence CFPs of livestock products. Transparency in methods and data are necessary to distinguish between actual differences and differences caused by methods and data. For harmonization, focus should be on methods to calculate NO_3^- leaching and emissions from LULUC. It is important to consider LUC in CFP studies of food, feed and bioenergy products.

Chapter 3

Evaluation of a feeding strategy to reduce greenhouse gas emissions from dairy farming: the level of analysis matters

C.E. van Middelaar[a], P.B.M. Berentsen[b], J. Dijkstra[c], and I.J.M. de Boer[a]

[a] Animal Production Systems group, Wageningen University, Wageningen, the Netherlands

[b] Business Economics group, Wageningen University, Wageningen, the Netherlands

[c] Animal Nutrition group, Wageningen University, Wageningen, the Netherlands

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Note: the opportunity is taken to correct errors in published results. Conclusions were unaffected by corrections.

Abstract

The dairy sector contributes to climate change through emission of greenhouse gases (GHGs), via mainly carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O). Replacing grass silage with maize silage is a feeding strategy to reduce enteric CH_4 emission. The effect of this strategy on GHG emissions can be analyzed at three different levels: animal, farm, and chain level. The level of analysis might affect results and conclusions, because the strategy affects not only enteric CH_4 emissions at animal level, but also other GHG emissions at farm and chain levels. The objective of this study was to determine if the level of analysis influences conclusions about the GHG reduction potential of increasing maize silage at the expense of grass and grass silage in a dairy cow's diet. First, we used a linear programming (LP, maximizing labor income) dairy farm model to define a typical Dutch dairy farm on sandy soils without a predefined feeding strategy (i.e. reference situation). Second, we combined mechanistic modeling of enteric fermentation and life cycle assessment to quantify GHG emissions at all three levels. Third, continuing from the diet derived in the reference situation, maize silage was increased by 1 kg DM per cow per day at the expense of grass (summer), or grass silage (winter). Next, the dairy farm model was used again to determine a new optimal farm plan including the feeding strategy, and GHGs were quantified again at the three levels. Finally, we compared GHG emissions at the different levels between the reference situation and the situation including the feeding strategy. We performed this analysis for a farm with an average intensity (13,430 kg milk/ha) and for a more intensive farm (14,788 kg milk/ha). Results show that the level of analysis strongly influences results and conclusions. At animal level, the strategy reduced annual emissions by 12.0 kg CO_2e per ton of fat-and-protein-corrected-milk (FPCM). Analysis at farm and chain level revealed first of all that the strategy is not feasible on the farm with an average intensity because this farm cannot reduce its grassland area because of compliance with the EU derogation regulation (a minimum of 70% grassland). This is reality for many Dutch dairy farms with an intensity up to the average. For the more intensive farm, that can reduce its area of grassland, annual emissions reduced by 11.1 kg CO_2e per ton FPCM at farm level, and 14.0 kg CO_2e per ton FPCM at chain level. Ploughing grassland into maize land, however, resulted in non-recurrent emissions of 860 kg CO_2e per ton FPCM. At farm and chain levels, therefore, the strategy does not immediately reduce GHG emissions as opposed to what results at animal level may suggest; at chain level it takes 61 years before annual emission reduction has paid off emissions from land use change.

1 Introduction

Environmental consequences of livestock production have received increasing attention over the last few decades. Among other concerns, attention increased because the livestock sector appears to cause approximately 18% of the global anthropogenic emissions of greenhouse gases (GHGs) (Steinfeld et al., 2006; De Vries and De Boer, 2010). Because this is currently one of the major environmental problems addressed, this study will focus solely on GHG emissions. Important GHGs related to livestock production are carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O). In dairy farming, CH_4 contributes approximately 52% to total GHG emissions at chain level, mostly caused by enteric fermentation processes within the cow (Gerber et al., 2010). For Dutch dairy farming, for example, per ton fat-and-protein-corrected milk (FPCM) approximately 15.4 kg enteric CH_4 is emitted (Bannink et al., 2011), which is 385 kg CO_2 equivalents (CO_2e) (Forster et al., 2007). The amount of enteric CH_4 is mainly related to the type and amount of feed (Dijkstra et al., 2007; Beauchemin et al., 2008; Ellis et al., 2008). To reduce GHG emissions from dairy farming, therefore, animal nutritionists propose feeding strategies that reduce enteric CH_4.

Several studies explored the CH_4 reduction potential of feeding strategies at animal level (Ellis et al., 2008; Grainger and Beauchemin, 2011). To predict the CH_4 reduction potential of a feeding strategy, mechanistic simulation models may be used, describing underlying mechanisms of enteric fermentation. A feeding strategy with potential to reduce enteric CH_4 emission, for example, is replacing grass silage with maize silage in a cow's diet (Mills et al., 2001; Beauchemin et al., 2008). Dijkstra et al. (2011) showed that replacing 50% of the grass silage with maize silage in a diet containing on average 30% concentrates and 70% grass silage, reduces enteric CH_4 levels by approximately 8%. Focusing at the animal level, this is a strategy with potential to reduce GHG emissions.

Literature shows, however, that dietary manipulation not only changes enteric CH_4 emission at animal level, but also other GHGs at farm and chain levels (Chianese et al., 2009; Kebreab et al., 2010). Replacing grass silage with maize silage, for example, will change the farm plan, i.e. part of the grassland will be ploughed into maize land. Ploughing grassland into maize land results in CO_2 and N_2O emissions, due to a change in soil carbon and nitrogen levels (Vellinga and Hoving., 2011; Van Middelaar et al., 2013a). Moreover, cultivating maize instead of grass requires different fertilization and land management, changing N_2O emissions from crop cultivation and emissions related to production of fertilizers (Schils et al., 2005; Basset-Mens et al., 2009a). Evaluation of a feeding strategy at animal level, therefore, might show different results than evaluation at farm level (including all on-farm processes), or evaluation at chain level (including also production of farm inputs), so level of analysis might matter.

Several studies address the importance of evaluating the effect of strategies to reduce GHG emissions at farm (Schils et al., 2006a; Rotz et al., 2010) or chain level (Lovett et al., 2006; Thomassen et al., 2008; Flysjö et al., 2011a; Kristensen et al., 2011). Roer et al. (2012) explored the influence of the level of analysis on environmental assessment of grain production in Norway, but to our knowledge, no study consistently determined this influence on the effect of feeding strategies to reduce GHG emissions from dairy farming.

The objective of this study was to determine if the level of analysis (i.e. animal, farm, chain) influences conclusions about the GHG reduction potential of increasing maize silage at the expense of grass and grass silage in a dairy cow's diet. A mechanistic model to predict enteric CH_4 emission at cow level is combined with a linear programming (LP) model to predict effects of a dietary change at farm level, and with life cycle assessment (LCA) to predict GHG emissions at chain level. Furthermore, a mechanistic model was used to determine GHG emissions related to ploughing grassland into maize land. The impact of the level of analysis is demonstrated using the cases of an average and a somewhat more intensive Dutch dairy farm on a sandy soil.

2 Methods

Quantifying GHG emissions from dairy farming at different levels requires definition of the system at each level. System boundaries for the different (interdependent) hierarchical levels are presented in Figure 1. The animal level focuses on processes within the dairy cow. The farm level focuses on processes on the farm, including milk production, manure management, and on-farm feed production. The chain level focuses on processes along the milk production chain up to the farm-gate, including on-farm processes, and production of farm inputs, such as synthetic fertilizers and concentrates. Processing stages after the farm gate were not analyzed, because these were assumed to be unaffected by the feeding strategy.

To determine changes in GHG emissions that result from increasing maize silage at the expense of grass and grass silage, information is needed about additional effects on diet and farm plan. For this reason, a dairy farm linear programming (LP) model was used, based on Berentsen and Giesen (1995). Based on the objective to maximize labor income of the farm family, this model optimizes the farm plan, including land use and diet. Dietary requirements, such as a minimum

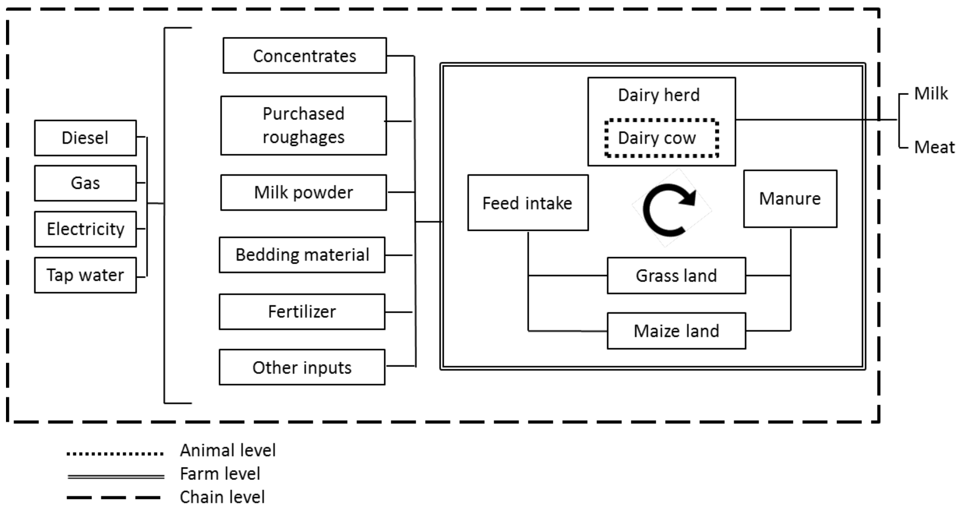

Figure 1. System boundaries for the three different aggregation levels.

amount of maize silage, can be included. We combined the LP model with several modeling techniques to calculate GHG emissions at the three levels. This resulted in a complete overview of all the consequences of the feeding strategy with respect to GHG emissions. In the next section the general set up of the LP model is explained. Subsequently, the models used to calculate GHG emissions at the different levels are described. Finally the set-up of the analysis is described.

2.1 Dairy farm LP model

The basic structure of the dairy farm LP model is described in Berentsen and Giesen (1995). The model includes all relevant activities and constraints that are common to Dutch dairy farms. The objective function maximizes labor income, i.e. gross returns minus variable and fixed costs. The solution procedure optimizes feeding, manure application, and land use, given the activities and constraints of the model. Important activities are (1) on-farm feed production, including production of maize silage and production of grass for grazing and silage making at different nitrogen (N) levels, (2) purchase of feed, including maize silage and a variety of concentrates with regard to protein content, (3) animal production, including dairy cows with young stock for replacement and sale, (4) manure application, (5) purchase and application of synthetic fertilizers, and (6) field operations, such as harvesting of grass and silage making.

The model distinguishes a summer period (182 days) and a winter period (183 days). Dietary options include grass from grazing, grass silage, maize silage, and three types of concentrates with different protein levels: standard, medium and high. All dietary options are available in winter and summer, except for fresh grass (only in summer). Table 1 shows the feed characteristics of the dietary options. Production of maize silage on-farm requires 150 kg N/ha and yields 86,900 MJ net energy for lactation (NE_L) per hectare, whereas production of grass depends on N fertilization level: 100 kg N yields 50,400 MJ NE_L/ha; 200 kg N yields 62,800 MJ; 300 kg N yields 71,800 MJ; and 400 kg N yields 77,300 MJ NE_L/ha. Cows belong to the Holstein Friesian breed and are assumed to calve in February. Female young stock is kept for yearly replacement of 29% of the dairy herd. Male calves and surplus female calves are sold at an age of two weeks (Berentsen and Giesen, 1995). Costs of farm inputs were updated according to current market prices, i.e. €39 per ton maize silage, €220 per ton concentrates with standard protein levels, €235 for medium protein levels and €290 for high protein levels (KWIN-AGV, 2009; CeHaVe, 2012). Milk was sold at a price of €310 per ton (Wageningen UR, 2011).

Table 1. Feed characteristics of the dietary options[1].

Dietary options	NE_L [2] (MJ/kg DM)	DVE [3] (g/kg DM)	OEB [4] (g/kg DM)	N (g/kg DM)	Fill value [5] (kg/kg)	NDF (g/kg DM)	Starch (g/kg DM)
Concentrates							
- standard protein	7.2	100	5.6	24.1	0.29-0.72	401	91
- medium protein	7.2	133	27.8	32.2	0.29-0.72	393	78
- high protein	7.2	200	83.3	48.3	0.29-0.72	299	73
Grazed grass							
- 100 kg N	6.6	93	6.0	27.3	0.93	453	0
- 200 kg N	6.7	97	18.8	29.9	0.93	449	0
- 300 kg N	6.8	101	33.5	32.5	0.93	445	0
- 400 kg N	6.9	104	50.3	35.1	0.93	441	0
Grass silage							
- 100 kg N	5.7	67	18.0	24.7	1.08	501	0
- 200 kg N	5.8	70	37.3	28.6	1.08	497	0
- 300 kg N	5.8	72	55.3	31.9	1.08	493	0
- 400 kg N	5.9	74	72.0	34.8	1.08	488	0
Maize silage	6.6	58	-36.0	13.4	1.02	403	342

[1] All dietary options are available in summer and winter, except for fresh grass (only in summer).
[2] Net energy for lactation.
[3] True protein digested in the small intestine according to Dutch standards (Tamminga et al., 1994).
[4] Rumen degradable protein balance according to Dutch standards (Tamminga et al., 1994).
[5] Fill value per kg feed expressed in kg of a standard reference feed (see Jarrige, 1988). The fill value of concentrates increases with an increase in concentrate intake.

Constraints of the model include fixed resources of the farm (e.g. land area, labor, milk quota, and housing facility), links between activities, and environmental policies. Examples of links between activities are the feed restrictions, which match on-farm feed production and purchased feed with animal requirements for energy and protein; and fertilizer requirements, which match the need for nutrients for grassland and arable land, with the available nutrients from manure and purchased fertilizers. Environmental policies include limits to the application of total N and phosphate fertilization, and to the application of N from animal manure on the land (DR Loket, 2012). The latter constraint is based on the European nitrate directive (Directive 91/676/EEC) and includes the derogation regulation, which is specific for a few countries in the EU that have a high proportion of grassland (EU, 2010). It prescribes that farms with at least 70% grassland can apply 250 kg N/ha originating from animal manure, instead of 170 kg N/ha that is prescribed for farms with less than 70% grassland.

2.2 Modeling GHG emissions

GHG emissions at different levels were calculated by summing up CO_2, CH_4, and N_2O emissions based on their equivalence factor in terms of CO_2 (100-year time horizon): 1 for CO_2, 25 for CH_4, and 298 for N_2O (Forster et al., 2007). Emissions were calculated per ton FPCM, i.e. the milk is corrected to a fat percentage of 4.0% and a protein content of 3.3% using the following formula (Product Board Animal Feed, 2008).

FPCM (kg) = Milk (kg) x [0.337 + 0.116 x Fat (%) + 0.06 x Protein (%)]

Economic allocation (i.e. allocation based on the relative economic value of the outputs; Guinée et al., 2002) was used to allocate emissions to the different outputs of the dairy farm (i.e. milk and meat). This method is used most commonly in LCA studies of livestock products (De Vries and De Boer, 2010). Of all emissions, 89% was allocated to the milk, whereas 11% was allocated to the meat. Also at chain level economic allocation was applied for emissions related to purchased feed products in case of multiple output systems.

Animal level

Analysis at animal level focuses on CH_4 emissions from enteric fermentation. Enteric CH_4 emissions were estimated with a mechanistic model originating from Dijkstra et al. (1992), which was modified and updated by Mills et al. (2001), and with volatile fatty acids stoichiometry from Bannink et al. (2006). The model simulates digestion, absorption, and outflow of nutrients in the rumen, small intestine and hindgut. Model inputs include DM intake, chemical composition of the dietary components, and rumen degradation characteristics of neutral detergent fiber (NDF), crude protein and starch. Degradation characteristics were derived from in situ experiments with nylon bags to incubate feed in the rumen of a cow. Based on non-linear equations the model describes interactions between feed substrates and ruminal microbes, resulting in the production of volatile fatty acids, microbial mass, and di-hydrogen (H_2). CH_4 production is estimated from the H_2 balance. Sources of H_2 include H_2 from production of acetate and butyrate, and from microbial growth with amino acid as N-source. Sinks of H_2 include H_2 used for production of propionate and valerate, for microbial growth with ammonia as N-source, and for biohydrogenation of unsaturated long chain fatty acids. The surplus H_2 is assumed to be completely converted into CH_4.

Replacing grass silage with maize silage reduces enteric CH_4 emission, because maize silage contains more starch and less fiber than grass silage (Table 1), which favors production of propionate over acetate and thus reduces CH_4 formation per unit fermented substrate (Mills et al., 2001; Beauchemin et al., 2008). Moreover, starch bypassing rumen fermentation is largely digested in the small intestine and no CH_4 is formed from this bypass starch, whereas fiber that is not fermented in the rumen cannot be digested in the small intestine (Dijkstra et al., 2011).

Farm level

Annual GHG emissions. Analysis at farm level focuses on GHG emissions related to processes on the farm, including enteric CH_4 emission, and emissions from manure, managed soils, and combustion of energy sources, such as diesel and gas. Methods to determine enteric CH_4 emission for dairy cows are described in the previous subsection. For young stock, CH_4 emissions were based on IPCC Tier 2 methods and default values, i.e. the average gross energy content of feed is assumed to be 18.45 MJ per kg DM, and 6.5% of the gross energy intake is converted to CH_4 (IPCC, 2006).

Besides enteric CH_4, the farm model includes CH_4 emissions from manure in stables and storages. Emissions factors are included in Appendix 3.a. Furthermore, both direct and indirect N_2O emissions are included. Direct N_2O emissions result from N application to the field, including N from manure, synthetic fertilizers, and crop residues. Indirect N_2O emissions result from N that is removed from the farm via leaching of NO_3^- and volatilization of NH_3 and NO_x (IPCC, 2006). Emission factors for calculating direct and indirect N_2O emissions, NO_3^- leaching, and NH_3 and NO_x volatilization are included in Appendix 3.a. On-farm CO_2 emissions relate to the combustion of gas and diesel for agricultural operations. Emission factors were based on Ecoinvent (2007), which combine emissions from production (off-farm), and combustion (on-farm) (Appendix 3.b). For the farm-level analysis, 50% of the emission factor presented in Appendix 3.b was assumed to be related to combustion.

Non-recurrent GHG emissions from on-farm land use change. In addition to the annual (i.e. recurrent) emissions that are described above, CO_2 and N_2O are emitted in a non-recurrent fashion when ploughing grassland into maize land, referred to as on-farm land use change (LUC). Grassland contains more soil organic matter, and therefore more carbon (C) and nitrogen (N) than maize land (Leifeld et al., 2005; Reijneveld et al., 2009). Part of the soil C and N that was sequestered in grassland is lost in the form of CO_2 and N_2O when changing to maize land (Vellinga and Hoving, 2011; Van Middelaar et al., 2013a). To quantify these emissions, the amounts of soil C and N in grassland and maize land before and after ploughing need to be assessed. To do so, we used the Introductory Carbon Balance Model (ICBM) (Andrén and Kätterer, 1997; Kätterer and Andrén, 1999), validated by Vellinga et al. (2004) for the Dutch situation. This model quantifies changes in soil C and N stocks over time. Stable C and N stock levels are reached after approximately 70 years. These stable levels are the maximum amount of C and N stocks in the soil when land use type and management remain unchanged.

For this study, we assumed that grassland was replaced with maize land using full tillage, and that no rotation was applied. Furthermore, grassland was assumed to be 60 years old (Vellinga et al., 2004) and renovated every 5 years since the last few decades (Aarts et al., 2002). Using the model, stock levels in 60 years old grassland were found to be 80.1 t C/ha and 5.3 t N/ha at the moment of ploughing. After ploughing the land into maize land, stock levels decreased to 40.7 t C/ha and 2.7 t N/ha after stabilization (i.e. after ploughing the land into maize land another 70 years were simulated). According to these results and IPCC (2006) emission factors for CO_2 and N_2O emissions from changes in soil C and N, total non-recurrent emissions during this 70 years period are 160 t CO_2e/ha.

In addition to these actual losses, we need to account for the loss of sequestration potential. This includes only CO_2 that would have been sequestered, if grassland would have remained grassland. To quantify this foregone CO_2 sequestration, the difference between C stock levels at the moment of ploughing (60 years old grassland), and after stabilization (70 years old grassland) were determined. After stabilization C-stock levels were found to be 81.0 t C/ha. The loss of sequestration potential is therefore 0.9 t C/ha (81.0-80.1), equivalent to 3 t CO_2e. The total amount of non-recurrent emissions caused by ploughing grassland into maize land were therefore estimated to be 163 t CO_2e/ha. Emissions from on-farm LUC were allocated to the different farm outputs (i.e. milk and meat) based on economic allocation. We did not amortize the emissions over a time period (e.g. 70 years of stabilization), but determined the total amount of non-recurrent emissions caused by implementing the strategy.

Chain level

Annual GHG emissions. Our chain level analysis focuses on GHG emissions related to all processes along the milk production chain up to the moment that milk leaves the farm gate (Figure 1). Life cycle assessment (LCA) was used to evaluate emissions at chain level. LCA is an internationally accepted and standardized method (ISO 14040, ISO 14041, ISO 14042, ISO 14043) that accounts for all emissions along a production chain, by evaluating the use of resources, and emission of pollutants of all chain stages (Bauman and Tillman, 2004; Rebitzer et al., 2004). Here we focused on GHG emissions only. Methods for GHG emissions at farm level are described in the previous subsections.

Farm inputs include synthetic fertilizer, pesticides, concentrates, maize silage, milk replacer, bedding material (i.e. saw dust), tap water, and energy (i.e. gas, diesel, and electricity). Emissions from the production of synthetic fertilizer, pesticides, tap water, and energy were based on Eco-invent (2007), and emissions from production of saw dust and milk replacer were based on Thomassen et al. (2009) (Appendix 3.b). Concentrate composition for standard, medium, and high protein concentrates were based on the average of the last three years (Nevedi 2009; 2010; 2011) and are presented in Table 2, together with the emissions per ingredient. Emissions per ingredient were based on Vellinga et al. (2012), and include emissions from the production of inputs (e.g. fertilizers, pesticides, machinery, energy), direct and indirect N_2O emissions from cultivation, CO_2 emissions from liming and urea fertilization, emissions from drying and processing, and emissions from transport in between stages, up to the farm gate. For the concentrates for young stock, an emission factor similar to the one for concentrates with standard protein levels was used. Inventory data to calculate emissions from production of purchased maize silage are in Appendix 3.c. Emission calculations were similar to the calculations that were used for on-farm production of maize silage (Appendix 3.a), because purchased maize silage was assumed to be produced in the Netherlands. Emissions were found to be 212 kg CO_2e/t DM.

Non-recurrent emissions from off-farm land use change. Production of concentrate ingredients (e.g. soybean meal) have been related to deforestation and other types of off-farm LUC (Galford et al., 2010; Macedo et al., 2012). Off-farm LUC in this study refers to transformation of forest land

Table 2. Concentrate composition for dairy cows and GHG emissions per ingredient and per composition.

Ingredients	Standard protein (%)	Medium protein (%)	High protein (%)	CO_2e (kg/t DM)
Citrus pulp	11.64	7.64	7.64	682
Barley	1.27	0.95	0.95	414
Soybean expeller	12.85	11.01	0.00	578
Molasses (cane)	2.10	2.10	2.10	562
Rapeseed expeller	0.32	2.17	0.00	643
Rye	2.92	1.84	1.84	470
Palm kernel expeller	19.33	19.33	19.33	502
Beet pulp	6.34	6.33	6.33	323
Maize	1.18	1.48	1.48	503
Wheat middling	9.51	2.62	2.62	259
Maize gluten feed	17.42	17.77	17.22	1279
Triticale	0.68	1.32	1.32	647
Rapeseed meal	4.52	11.34	0.00	572
Soybean meal protected	0.00	0.00	28.55	623
Maize distillers	4.81	4.97	4.97	798
Wheat distillers	1.53	2.50	2.50	798
Urea	0.00	0.00	1.70	3878
Other	3.59	6.63	1.44	variable
GHG emissions (CO_2e)	(kg/t DM)	(kg/t DM)	(kg/t DM)	
Ingredients	653	675	753	
Feed mill	54	54	54	
Transport to farm	11	11	11	
Total	719	741	819	

Compositions are based on Nevedi (2009; 2010; 2011). GHG emissions are based on Vellinga et al. (2012).

and scrubland into cropland used for the production of purchased feed products, and is found to be an important source of GHG emissions. Calculation methods for LUC emissions, however, show high uncertainty and variability (Flysjö et al., 2011a; Van Middelaar et al., 2013a). We therefore used two different methods, and reported these emissions separately from other emissions. The first method focuses on direct LUC within a country or region (Jungbluth et al. 2007; Prudencio da Silva et al. 2010). Soybean meal and palm kernel expeller were the only two ingredients related to direct LUC. For soybean meal, 1% of the soy from Central West Brazil was assumed to contribute to deforestation of tropical forest, and 3.4% to conversion of scrubland (Prudencio da Silva et al., 2010). For palm kernel expeller, 100% of the palm fruit from Malaysia was assumed to contribute to deforestation of tropical forest (Jungbluth et al., 2007). Emissions per ha LUC were based on IPCC (2006), i.e. 825 t CO_2e per ha of tropical forest and 297 t CO_2e per ha of scrubland for soybean meal, and 496 t CO_2e per ha of forest for palm kernel expeller (Van Middelaar et al., 2013a). The second method is based on Audsley et al. (2009) and does not differentiate between products that are directly or indirectly related to LUC. Because of globalization of food and feed markets, every ha of land used for commercial production is held responsible for deforestation. Total GHG emissions from deforestation at world level for the year 2004 were therefore divided by the total amount of land used for agricultural production, resulting in one emission factor of 1.43 t CO_2e/ha of land. In case of multiple output systems, i.e. soybean meal derives from processing soybeans into oil, meal and expeller, LUC emissions were allocated to the feed ingredients based on economic allocation, i.e. similar to the emissions from cultivation (Vellinga et al., 2012). LUC emissions per farm input for the two methods are included in Appendix 3.b.

2.3 Set up of the analysis

Effects of the maize silage strategy were calculated for an average Dutch dairy farm, and a more intensive Dutch dairy farm, both on sandy soil. The average farm has 44.9 ha of land, housing facility for 76 dairy cows with young stock, and a milk quota of 603 tons of milk per year. The more intensive farm has the same land area but housing facility for 84 dairy cows and a higher milk quota, i.e. 110% compared to the average farm. Milk production per cow was assumed to be 7968 kg/year (4.39% fat and 3.52% protein) for both farms. Data for the average farm were based on the Farm Accountancy Data Network (FADN) of the Agricultural Economics Research Institute from the Netherlands (FADN, 2012). The more intensive farm was evaluated because over time dairy farms appear to become more intensive. Abolition of the EU milk quota system in 2015 might accelerate this process (Louhichi et al., 2010).

The LP model was used for economic optimization of the average and the intensive farm before (reference situation) and after implementing the maize silage strategy. For the reference situations one additional feeding constraint was included. We assumed that only limited grazing was applied, i.e. the cows were pastured during the day and housed during the night. The maximum fresh grass intake in summer was therefore set at 12 kg DM/cow/day, corresponding to the maximum for fresh grass intake of cows that graze during day times, but are housed inside during the night (Taweel et al., 2004; Abrahamse et al., 2009). Economic optimization resulted in diets and farm plan for the reference situation. Subsequently, a requirement for maize silage was introduced, being 1 kg DM/cow/day greater than the amount of maize silage in the reference situation. Economic optimization was used again to determine the complete diets and the farm plan for the situation after implementing the strategy.

In the farm model, milk yield per cow is kept the same both before and after implementing the feeding strategy. This means that the model will adjust the diets in the situation with the feeding strategy such that energy and protein requirements are exactly met. Replacing grass silage with maize silage could be expected to slightly increase milk yield, because maize silage has a higher energy and protein content than grass silage (Abrahamse et al., 2009). On the other hand, replacing fresh grass with maize silage might slightly reduce milk yield.

A sensitivity analysis was performed to assess the effect of changes in parameters that contain high uncertainty and have a high impact on the outcome of the study. These were emissions from production of concentrates and from ploughing grassland into maize land. For concentrates, assumptions on input parameters such as yield, fertilization, and energy used for processing per ingredient affect emissions. Based on differences found in literature (Nguyen et al., 2012; Van Middelaar et al., 2013a; Vellinga et al., 2012) emissions per ingredient (Table 2) were changed by 25%. For ploughing grassland into maize land, emissions depend on assumptions on soil C and N stock differences between grassland and maize land. Based on differences found in literature (Kuikman et al., 2003; Vellinga et al., 2004; Leifeld et al., 2005; IPCC, 2006; Reijneveld et al., 2009; Sonneveld and Van Den Akker, 2011), non-recurrent emissions per ha were changed by 30%. Emissions from off-farm LUC (i.e. deforestation related to the production of concentrates) were not included in the sensitivity analysis because their uncertainty is reflected by the two different methods that were used.

3 Results and discussion

3.1 Diets and farm plan

Table 3 shows the diets and farm plan for the average and the intensive farm, before (reference) and after implementing the maize silage strategy. For the reference situation the following results apply. In summer, both the average and intensive farm used the maximum amount of 12 kg DM fresh grass/cow/day, because grazing is the cheapest way of feeding. Subsequently, requirements for energy, intake capacity, and rumen degradable protein balance were met by maximizing the amount of maize silage in combination with medium and high protein concentrates. In winter, a

Table 3. Diets and farm plan for an average and an intensive Dutch dairy farm, before (reference) and after (strategy) implementing the maize silage strategy.

		Average farm		Intensive farm	
		Reference	Strategy	Reference	Strategy
Diet dairy cows - summer period (kg DM/cow/day)					
Grass (grazed)		12.00	10.65	12.00	10.65
Maize silage		5.30	6.30	5.30	6.30
Concentrates total		2.01	2.39	2.01	2.39
- standard protein		-	-	-	-
- medium protein		1.54	0.77	1.54	0.77
- high protein		0.47	1.62	0.47	1.62
Diet is restricted by [1]		E,I,R	E,I,R	E,I,R	E,I,R
Diet dairy cows - winter period (kg DM/cow/day)					
Grass silage		5.50	4.50	5.50	4.50
Maize silage		5.60	6.60	5.60	6.60
Concentrates total		5.41	5.28	5.41	5.28
- standard protein		-	-	-	-
- medium protein		5.41	3.68	5.41	3.68
- high protein		-	1.60	-	1.60
Diet is restricted by [1]		E,I,R	E,R	E,I,R	E,R
Farm plan					
Dairy cows	n	75.7	75.7	83.3	83.3
Grassland 250N	ha	34.4	30.6	37.9	33.7
Maize land	ha	10.5	13.1	7.0	11.2
Synthetic fertilizer	kg N/ha	131.2	116.0	134.3	122.0
	kg P_2O_5/ha	14.0	12.2	7.4	8.8
Purchased maize silage	ha	5.3	5.0	10.4	8.7
Purchased concentrates	t DM	108.8	112.2	119.7	123.4
Labor income	€	32,435	29,341	34,969	32,664

[1] The diet has to meet a minimum requirement for energy, rumen degradable protein balance, and true protein digested in the small intestine without exceeding the intake capacity. Different feed ingredients have different feed characteristics. When the diet is restricted by one or more of these requirements it means that it cannot select the ingredients based on cost price only. E=energy requirements; I=intake capacity; R=rumen degradable protein balance.

combination of grass silage, maize silage, and concentrates were fed such that requirements for energy , intake capacity, and rumen degradable protein balance were exactly met in both farms.

For the farm plan, both the average and the intensive farm produced the maximum amount of milk as determined by the milk quota. In the reference situation, the average farm had 77% grassland and 23% maize land. The intensive farm had 84% grassland and 16% maize land, because the higher number of cows requires a larger area of grassland. The amount of purchased feed was higher for the intensive than for the average farm, especially for maize silage, because less maize silage could be produced on farm. Labor income per year was €32,435 for the average and €34,969 for the intensive farm.

Implementing the maize silage strategy resulted on both farms in a decrease of 1.35 kg DM of fresh grass per cow per day in summer. Replacement of grass with concentrates occurred, because in the reference situation energy requirements and intake capacity were restricting (Table 3). Because maize silage has a higher fill value and lower energy content per kg DM than grass, exchanging grass for maize silage necessitates an additional exchange of grass for concentrates to fulfill the energy requirements within the limiting intake capacity. Part of the medium protein concentrates were exchanged for high protein concentrates to compensate for the negative rumen degradable protein balance in maize silage (Table 1). For the winter diet, in both farms maize silage was increased at the expense of grass silage. The lower fill value and higher energy content of maize silage compared with grass silage decreased the amount of concentrates for both farms. Again, high protein concentrates were fed to compensate for the negative rumen degradable protein balance of maize silage.

For the farm plan of the average farm, increasing maize silage at the expense of grass and grass silage decreased the area of grassland by 3.8 ha, while maize land only increased by 2.6 ha. Purchased maize silage decreased from 5.3 to 5.0 ha per year (Table 3). Thus, introducing the strategy resulted in a situation where, at the one hand, not all farm land is used, but, at the other hand, maize silage is purchased. If the farm would grow more maize silage on-farm, the grassland share of the farm would be below 70%. Following the derogation regulation, this would mean that the maximum amount of N from manure that can be applied on the farm drops from 250 to 170 kg/ha. As a result, a substantial amount of manure would need to be removed from the farm, which is more expensive than leaving a relatively small area fallow to keep the derogation option. These results show that increasing maize silage at the expense of grass and grass silage is not a realistic option for Dutch dairy farms that comply almost or exactly with the 70% grassland requirement of the derogation regulation. Evaluation at farm level is needed to reveal these conclusions. Because of the unrealistic situation, the strategy will not be evaluated further for the average farm.

For the intensive farm, implementing the strategy did not result in a decrease of total farm area. The intensive farm could replace grassland with maize land without exceeding the derogation regulation and 4.2 ha were replaced. In addition, the amount of purchased maize silage decreased, because maize land has a higher DM yield per ha than grassland, resulting in an increase in total on-farm roughage production. The amount of synthetic fertilizer N decreased, because maize land uses less N fertilizer than grassland. Furthermore, labor income decreased by approximately 7%. This is mainly caused by the difference in costs of producing grass and grass silage compared to maize silage. The decrease in costs of purchased maize silage is leveled out by the increase in costs for concentrates. Costs of concentrates increased because of the price difference between medium and high protein concentrates.

3.2 GHG emissions

Animal level

At animal level, GHG emission in the reference situation was 370.6 kg CO_2e/t FPCM (Table 4). Expressed in grams CH_4/cow/day, emissions were 421 in summer and 351 in winter, averaging at 16.7 g CH_4/kg FPCM. This is higher than the average of 15.4 g CH_4/kg FPCM reported by Bannink et al. (2011). This difference is caused by the difference in diet. Our diet contained relatively more roughage and more fresh grass, resulting in higher CH_4 emissions. The difference between emissions in summer and winter is largely explained by the difference in DM intake between these seasons (viz., 14% higher intake in summer than in winter).

Compared to the reference situation, the strategy reduced GHG emissions by 12.0 kg CO_2e/t FPCM. The reduction potential of the strategy at animal level is 3.2%. This results from a 2.1% reduction of enteric CH_4 in summer and 4.6% in winter. In winter the reduction was larger because DM intake decreased, while in summer it increased (Table 3). Dijkstra et al. (2011) found a total reduction of 8% when exchanging 50% of the grass silage for maize silage in a mixed diet containing on average 70% grass silage and 30% concentrates. The more grass silage is exchanged, the higher the CH_4 reduction (Mills et al., 2001), which explains the difference between our results and results reported by Dijkstra et al. (2011).

Farm level

Annual GHG emissions. At farm level, GHG emission in the reference situation was 669.0 kg CO_2e/t FPCM (Table 4). Implementing the maize silage strategy slightly increased emissions from manure (+0.3 kg CO_2e/t FPCM) because of an increased N content in manure. The strategy, however, significantly decreased emissions from grassland (-10.2 kg CO_2e/t FPCM), because less grass is produced, whereas it increased emissions from maize land (+10.9 kg CO_2e/t FPCM). The total annual emission from on-farm feed production remained similar, as grassland has a higher

Table 4. GHG emissions for the intensive Dutch dairy farm (reference), and effects of increasing maize silage with 1 kg DM/cow/day at the expense of grass and grass silage on annual and non-recurrent emissions at three interdependent hierarchical levels[1], in kg CO_2e/t FPCM.

Level		Reference	Effect of strategy: Annual emissions	Effect of strategy: Non-recurrent emissions
Chain level / Farm level / Animal level	*Animal level emission*			
	Enteric CH_4 emission dairy cows	370.6	-12.0	
	Total animal level	370.6	-12.0	
	Additional farm level emissions			
	Enteric CH_4 emission young stock	79.1		
	Manure	116.1	+0.3	
	On-farm feed production			
	Grassland	84.7	-10.2	
	Maize land	18.5	+10.9	
	Ploughing grassland for maize land			+860
	Total farm level	669.0	-11.1	+860
	Additional chain level emissions			
	Concentrates dairy cows	106.2	+7.6	
	Concentrates young stock	8.3		
	Purchased roughages	35.3	-5.6	
	Milk replacer	2.4		
	Bedding material	3.6		
	Synthetic fertilizer	66.7	-5.9	
	Other inputs	17.2	+0.9	
	Total chain level	908.8	-14.0	+860
	Off-farm LUC emissions			
	Method 1, direct LUC	16.5	+2.2	
	Total chain level incl. method 1	925.2	-11.8	+860
	Method 2, all land equally responsible	131.4	-0.3	
	Total chain level incl. method 2	1040.1	-14.3	+860

[1] Each level includes emissions from the previous level(s).

fertilization rate (250 kg N/ha) than maize land (150 kg N/ha), but a lower N_2O emission factor per m^3 manure applied (Appendix 3.a). Based on annual emissions only, the reduction potential of the feeding strategy at farm level is 1.7%.

Non-recurrent emissions from on-farm land use change. Replacing grassland with maize land that resulted from the strategy, caused non-recurrent emissions of 860 kg CO_2e/t FPCM. From a farm-perspective, the carbon payback time of the strategy, i.e. the years of mitigation that are needed before LUC emissions are compensated, is 78 years.

Several assumptions were made that influence the level of non-recurrent emissions. Firstly, assumptions on grass yield and maize silage yield per ha determine the amount of grass land that is changed into maize land. Maize silage yield was based on the Handbook Quantitative Information on Animal Husbandry (KWIN-V, 2008), but was at the lower end when compared to national statistics (CBS, 2013). When maize yield per ha increases, less grassland has to be ploughed, resulting in lower non-recurrent emissions and a lower carbon payback time. Secondly, assumptions on grassland age and renovation frequency influence the amount of C and N stored in grassland, and therefore emissions from ploughing to replace grassland by maize land. Because foregone C sequestration of grassland has been included, the age of grassland does not affect CO_2 emissions, but only N_2O emissions. Because N_2O emissions contributed only 10% to the total emissions, the assumption on the age of grassland does not affect results much. Assumptions on renovation frequency have a stronger effect. Renovation was assumed to be applied only during the last few decades when storage capacity was almost reached. Soil C and N stock levels were in line with results found by Vellinga et al. (2004). Model simulations, however, show that when renovation would have been applied from the beginning, C storage under grassland would have been approximately 20% lower, and non-recurrent emission would reduce from 163 t CO_2e/ha to 110 t CO_2e/ha, further reducing carbon payback time from 78 to 53 years. Thirdly, we assumed no rotation of grassland and maize land, resulting in a non-recurrent emission of 163 t CO_2/ha. In a situation with rotation, emissions per ha will be lower, because grass/maize in rotation sequesters more C and N in the soil than cultivating maize only (Vellinga and Hoving, 2011). At farm level, however, Vellinga and Hoving (2011) showed that non-recurrent emissions will be higher when rotation is applied, because more ha of grassland are changed into maize land on a regular basis, reducing the total sequestration potential of the farm area. In addition, annual N_2O emissions are higher in case of rotation. No rotation, therefore, is assumed to be the most optimal scenario to reduce GHG emissions in situations of full tillage. In case of strip tillage, or no tillage, non-recurrent emissions are expected to be lower (Vellinga and Hoving, 2011).

Chain level

Annual GHG emissions. At chain level, GHG emission in the reference situation was 908.8 kg CO_2e/t FPCM (Table 4), which is at the lower end of the range of results that is found in literature (Basset-Mens et al., 2009b; Van Middelaar et al., 2011; O'Brien et al., 2012). Two factors have contributed to this. First, we used a model farm, which is comparable to the more efficient farms that can be found in practice. Second, we used a mechanistic model to calculate enteric CH_4

emissions, whereas most other studies used IPCC Tier 2 methods that generally overestimate enteric CH_4 emissions (Kebreab et al., 2008; Alemu et al., 2011).

Implementing the strategy increased emissions from concentrate production (+7.6 kg CO_2e/t FPCM). This partly resulted from an increased use of concentrates (Table 3), and partly from a change in type of concentrates (i.e. less medium protein concentrates and more high protein concentrates, see Table 2 for emission factors). Reduced emissions were related to a decrease in the amount of purchased maize silage (-5.6 kg CO_2e/t FPCM), and a decrease in synthetic fertilizer use (-5.9 kg CO_2e/t FPCM). Emissions from other inputs slightly increased (+0.9 kg CO_2e/t FPCM) because on-farm production of maize silage uses more machinery and diesel than the combination of grass and grass silage. Based on annual emissions, the feeding strategy reduced emissions by 1.5% at chain level.

Non-recurrent emissions from on-farm land use change. Including non-recurrent emissions from on-farm LUC, the carbon payback time of the strategy at chain level is 61 years. Thus, the strategy does not immediately reduce GHG emissions as opposed to what results at animal level may suggest. Using a 100 year time horizon to calculate emissions, however, we conclude that the carbon payback time falls well within this time frame and the strategy offers potential to reduce GHG emissions.

Vellinga and Hoving (2011) also considered the consequences of increasing maize silage in a dairy cow's diet at chain level GHG emissions, but did not explicitly replace grass and grass silage. The set-up of the analysis was also different as they did not used LP to determine the additional consequences of the strategy. The overall conclusion that it takes up to several decades before annual emission reduction has paid off emissions from on-farm land use change, however, was the same. Vellinga and Hoving (2011) found a carbon payback time of 60 years.

Non recurrent emissions from off-farm land use change. Including emissions from off-farm LUC, e.g. deforestation related to the production of purchased feed, increased GHG emissions in the reference situation by 16.5 kg CO_2e/t FPCM when using method 1 (direct LUC), and by 131.4 CO_2e/t FPCM when using method 2 (i.e. all land responsible). Implementing the strategy increased LUC emissions by 2.2 kg CO_2e/t FPCM for method 1, and decreased LUC emissions by 0.3 kg CO_2e/t FPCM for method 2. Hence, including off-farm LUC emissions resulted in a carbon payback time of 73 years when using method 1, and 60 years when using method 2.

Including off-farm LUC emissions did not change the overall outcome of the study drastically. Results, however, show that methods for off-farm LUC emissions lack consistency and that the method chosen can influence results. In addition, emission calculations contain high uncertainty (Flysjö et al., 2012; Van Middelaar et al., 2013a). Direct LUC emissions, for example, can easily change when assumptions on deforestation rate or economic value of the feed ingredients changes. Compared to Flysjö et al. (2012), who included LUC emissions in GHG calculations for Swedish milk production, for example, we used much lower deforestation rates for soybean meal resulting in lower direct LUC emissions per t FPCM, i.e. approximately one fifth of the emissions that Flysjö et al. (2012) found. This shows that results highly depend on underlying assumptions.

Method 2 uses a single emission factor per ha of land. The decrease in LUC emissions when implementing the strategy reflects the decrease in purchased maize silage (1.6 ha, Table 3), partly compensated by an increase in land use due to a change in concentrate use. In line with emission calculations for on-farm LUC (i.e. ploughing grassland for maize land), we could argue that the 1.6 ha that is saved following a reduction in purchased maize silage, can be changed into grassland or forest land. If we allocate the C-sequestration potential of this land use change to the strategy, non-recurrent emissions reduce by 320 kg CO_2e/t FPCM (changing to grassland, results based on calculations with the ICBM model), or 475 kg CO_2e/t FPCM (changing to natural vegetation, based on IPCC, 2006). Both scenarios, however, are not very likely. It is more likely that the land remains in use of agricultural production and does not result in C-sequestration. In addition, when using this approach we should also consider the effects of an increase in land use needed for concentrate production (representing 1.4 ha of land) in a similar way. The complexity of the global feed market makes it difficult to determine the exact consequences of a change in diet on land use and off-farm LUC. Overall, in this case method 2 seems to be the right way to credit the overall reduction in off-farm land use resulting from the strategy.

3.3 Sensitivity analysis

Results of the sensitivity analysis are presented in Table 5. The average carbon payback time of the strategy at chain level is 61 years. Changing emissions from concentrate production resulted in a carbon payback time of minimal 44 and maximal 78 years, and changing emissions from grassland ploughing in minimal 43 and maximal 80 years. When changing both, the minimum carbon payback time is 31 years and the maximum 101.

To show maximum and minimum values following a change in emissions from concentrate production, we increased emissions from ingredients used mostly in the reference situation while decreasing emissions from ingredients used mostly for the strategy (pro strategy), and vice versa

Table 5. Effects of a change in GHG emissions from concentrate production, ploughing grassland into maize land, and both, on annual emission reduction, non-recurrent emissions and carbon payback time of the maize silage strategy.

	Change in emission factors	Annual emission reduction	Non-recurrent emissions	Carbon payback time
		kg CO_2e/t FPCM	kg CO_2e/t FPCM	years
Average	-	-14.0	860	61
Concentrate production	pro strategy[1]	-19.5	860	44
Concentrate production	con strategy[1]	-11.1	860	78
Ploughing grassland	-30%	-14.0	602	43
Ploughing grassland	+30%	-14.0	1118	80
Conc. prod. & ploughing	pro strategy	-19.5	602	31
Conc. prod. & ploughing	con strategy	-11.1	1118	101

[1] Emissions per ingredient were changed by plus or minus 25%. Pro strategy: annual emission reduction was maximized by increasing emissions from ingredient used mostly in the reference situation, and decreasing emissions from ingredients used mostly for the strategy. Con strategy: emission reduction was minimized by doing the opposite.

(con strategy). In addition to a change in emissions per ingredient, emissions from concentrate production can also change due to a change in concentrate composition. We did not include the effect of such a change, because concentrate compositions very much depend on market prices of the single ingredients and therefore it is unlikely that the different concentrate types will show completely different compositions. Hence, a change in composition will affect emissions per t FPCM for both the reference situation and the strategy, but not so much the effect of the strategy.

Emissions from grassland ploughing depend on the difference in C and N stocks in maize land compared to grassland. We found a difference of minus 40 t C/ha and 3 t N/ha, which seems high compared to other studies that quantified soil C stocks in grassland and arable land based on soil surveys (Kuikman et al., 2003; Leifeld et al., 2005; Reijneveld et al., 2009). This could indicate that in practice, emissions from ploughing grassland into maize land are lower than reported here. Variation between studies, however, is large and also dependent on region, soil type and management practices (Reijneveld et al., 2009). Frequent renovation of permanent grassland, for example, reduces the C sequestration potential of the land, and hence the carbon payback time of the strategy by decreasing the difference between C and N stock levels in maize land compared to grassland. Moreover, grassland renovation in intensive dairy production systems may help to avoid a decline in grass and animal productivity (Hopkins et al., 1990).

The large range of possible outcomes that is presented by this sensitivity analysis shows how greatly results depend on assumptions about emissions related to feed production and grassland ploughing. This emphasizes the importance of correct assessment of production parameters and emission factors. Moreover, the large range represents also the variation that can occur in practice because of variation between production circumstances.

4 General discussion

Several methods and databases are available to quantify GHG emissions from dairy farming. We selected the methods that were assumed to be most accurate. These were not always the methods that are used most often. Enteric CH_4 emissions, for example, are often based on IPCC Tier 2 (IPCC, 2006). IPCC Tier 2 estimates enteric CH_4 emission based on empirical relations between the gross energy content of the diet and CH_4 production, i.e. 6.5% of the gross energy is converted into CH_4. Kebreab et al. (2008) and Alemu et al. (2011) showed that the mechanistic model has a much lower prediction error than IPCC Tier 2, when comparing the results of both methods to an independent dataset of CH_4 production in dairy cattle. Furthermore, errors in the mechanistic model are almost completely related to random errors, whereas IPCC Tier 2 significantly overestimates CH_4 production, and bias has a much higher contribution to the total prediction error (Kebreab et al., 2008; Alemu et al., 2011). Another example relates to emissions from the production of purchased feed products, which are often based on Eco-invent (Eco-invent, 2007). Eco-invent provides data on emissions from cultivation and processing of ingredients, and from off-farm LUC. We did not use Eco-invent to quantify emissions from off-farm LUC, for example, because the LUC emissions in Eco-invent only include GHG emissions from burning of above ground biomass and changes in soil C, which results in an underestimation of LUC emissions that also involves other GHG emissions (Van Middelaar et al., 2013a).

The strategy was applied to an average farm and a farm with a 10% higher intensity. The more intensive farm was included to show the potential of the feeding strategy in a situation where replacement of grassland with maize land is possible. We could have chosen a farm with a 20% or 30% higher intensity, but the effect of the feeding strategy would not have been different.

We increased maize silage by 1 kg DM/cow/day at the expense of grass and grass silage. The more grass and grass silage is exchanged for maize, the larger the reduction of enteric CH_4 (Mills et al., 2001). By exchanging 2 instead of 1 kg DM of grass or grass silage for maize silage, the effect on enteric CH_4 emission is approximately twice as high. The same accounts for the effect on other annual and non-recurrent emissions along the chain. This means that increasing the amount of maize silage with more than 1 kg DM/cow/day does not change the carbon payback time of the strategy, but the annual emission reduction will be larger after GHG emissions from on-farm LUC have been paid off. Increasing maize silage with more than 1 kg DM/day, however, is only possible as long as the area of grassland on the farm is at least 70%, which is the minimum requirement for grassland to comply with the derogation regulation. This feeding strategy, therefore, offers possibly more potential to reduce GHG emissions for countries that do not have a derogation regulation.

Implementing the strategy on the more intensive farm reduced labor income with approximately 7%. Without any further incentive to reduce GHG emissions, this decrease in income will prevent the farmer from implementing the strategy. In the future, policy constraints, or an increasing demand for more sustainable products, might provide an extra stimulus.

In 2015 the milk quota system will be abolished. This might change the Dutch dairy system (Louhichi et al., 2010). However, the conclusions of this paper still apply after abolition of the milk quota system. Given the growing conditions for crop production in the Netherlands, such as the length of the growing season, grass will remain the favorite type of forage. This means that replacing grass and grass silage with maize silage will still be an option to decrease GHG emissions. Due to environmental policies such as the European nitrate directive, intensification of Dutch dairy farms will be limited (EL&I, 2009). Effects of implementing the maize silage strategy, therefore, will not differ much between a situation with and without a milk quota.

5 Conclusions

The study showed that conclusions about the potential of a feeding strategy to reduce GHG emissions strongly depend on the level of analysis. At animal level, increasing maize silage at the expense of grass and grass silage in a dairy cow's diet, is a promising strategy with an immediate effect on GHG emissions. Analysis at farm and chain level reveals that the strategy is not feasible on farms that cannot further reduce their grassland area because of compliance with the EU derogation regulation. For more intensive farms that can reduce their grassland area, it takes 61 years at chain level, before annual emission reduction has paid off emissions from land use change. Results show the importance of a chain level analysis of strategies that reduce GHG emissions at animal level.

Chapter 4

Cost-effectiveness of feeding strategies to reduce greenhouse gas emissions from dairy farming

C.E. van Middelaar[a], J. Dijkstra[b], P.B.M. Berentsen[c], and I.J.M. de Boer[a]

[a] Animal Production Systems group, Wageningen University, Wageningen, the Netherlands

[b] Animal Nutrition group, Wageningen University, Wageningen, the Netherlands

[c] Business Economics group, Wageningen University, Wageningen, the Netherlands

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Abstract

The objective of this paper was to evaluate the cost-effectiveness of three feeding strategies to reduce enteric CH_4 production in dairy cows, by calculating the effect on labor income at farm level and on greenhouse gas (GHG) emissions at chain level (i.e. from production of farm inputs up to the farm gate). Strategies included were: dietary supplementation of an extruded linseed product (56% linseed; 1 kg/cow per day in summer and 2 kg/cow per day in winter), dietary supplementation of a nitrate source (75% nitrate; 1% of DM intake), and reducing the maturity stage of grass and grass silage (grazing at 1400 instead of 1700 kg DM/ha and harvesting at 3000 instead of 3500 kg DM/ha). A dairy farm linear programing model was used to define an average Dutch dairy farm on sandy soil without a predefined feeding strategy (reference situation). Subsequently, one of the three feeding strategies was implemented and the model was optimized again to determine the new economically optimal farm situation. Enteric CH_4 production in the reference situation and after implementing the strategies was calculated based on a mechanistic model for enteric CH_4, and empirical formulas explaining the impact of fat and nitrate supplementation on enteric CH_4 production. Other GHG emissions along the chain were calculated using life cycle assessment. Total GHG emissions in the reference situation added up to 840 kg CO_2e per ton fat-and-protein-corrected-milk (FPCM), and yearly labor income to €42,605. Supplementation of an extruded linseed product reduced emissions by 9 kg CO_2e/t FPCM, and labor income by €16,041; supplementation of a dietary nitrate source reduced emissions by 32 kg CO_2e/t FPCM, and labor income by €5,463; reducing the maturity stage of grass and grass silage reduced emissions by 11 kg CO_2e/t FPCM, and labor income by €463. Of all three strategies, reducing grass maturity was most cost-effective (€57/t CO_2e compared to €241/t CO_2e for nitrate and €2594/t CO_2e for linseed), and has the highest potential to be used in practice because additional costs are low.

1 Introduction

Methane (CH_4) production from enteric fermentation in dairy cows is not only an energy loss for the animal (i.e. about 6% of the gross energy intake is lost as CH_4), but also an important contributor to greenhouse gas (GHG) emissions (Ellis et al., 2008). Enteric CH_4 is responsible for about 50% of the total amount of GHG emissions along the production chain of milk, whereas other important GHG emissions are the emission of carbon dioxide (CO_2) and nitrous oxide (N_2O) (Hörtenhuber et al., 2010). Reducing enteric CH_4 production, therefore, is seen as an effective way to reduce GHG emissions from milk production.

Enteric CH_4 derives from microbial fermentation of feed substrates in the rumen (92%) and large intestine (8%) (Bannink et al., 2011). The production of CH_4 is influenced by dietary factors, such as type and amount of feed; animal factors, such as milk yield and genetic traits; and environmental factors, such as temperature (Kebreab et al., 2006a; Hristov et al., 2013a). Examples of feeding strategies that have been proposed to reduce enteric CH_4 are: dietary supplementation of fatty acids, dietary supplementation of nitrate, and reducing the maturity stage of grass and grass silage (Sterk et al., 2010; Van Zijderveld et al., 2011; Brask et al., 2013). Martin et al. (2008) showed that supplementation of extruded linseed to achieve a dietary fat content of 5.7% reduced enteric CH_4 from 19.3 g/kg FCM to 14.8 g/kg FCM. Supplementation of a nitrate source (75% nitrate; 2.1% of dietary DM) was found to reduce enteric CH_4 from 13.5 g/kg milk to 11.6 g/kg milk (Van Zijderveld et al., 2011). In a study on the effect of grass maturity on enteric CH_4, Brask et al. (2013) showed that feeding grass silage from an early cut resulted in lower CH_4 production/kg ECM than grass silage that was harvested three weeks later (i.e. 15.6 g/kg ECM compared to 17.8 g/kg ECM, own calculation based on milk yield and CH_4 production per day given by Brask et al. (2013)).

Implementing a feeding strategy not only affects enteric CH_4 production, but also other GHG emissions along the chain (Van Middelaar et al., 2013b; Williams et al., 2014). To analyze if the strategy results in a net reduction in GHG emission at chain level, i.e. from feed production to milk harvesting, an integrative approach such as life cycle assessment (LCA) is needed (Van Middelaar et al., 2013b). To our knowledge, so far no study examined the effect of mentioned feeding strategies on GHG emissions along the chain.

Insight into the economic effects of a strategy is required to determine if the strategy has potential to be used in practice (Hristov et al., 2013b). Farmers are more willing to implement strategies when the economic effects are positive or when negative effects are small (Vellinga et al., 2011).

Cottle et al. (2011) conclude that most strategies to reduce enteric CH_4 from ruminants are currently not profitable, which hampers their implementation. The cost-effectiveness of strategies provides insight into the economic effect per unit of GHG emission reduced.

The objective of this study was to analyze the cost-effectiveness of three feeding strategies to reduce CH_4 production, by calculating the economic effect at farm level per unit of net reduction in GHG emissions at chain level. Strategies evaluated were dietary supplementation of extruded linseed, dietary supplementation of nitrate, and reducing the maturity stage of grass and grass silage. To determine changes in labor income at farm level and in GHG emissions at chain level, we used a dairy farm linear programing (LP) model, a mechanistic model to predict enteric CH_4 production of dairy cattle, and LCA. Strategies were evaluated for an average Dutch dairy farm on sandy soils.

2 Methods

2.1 Dairy farm LP model

A dairy farm LP model based on Berentsen and Giesen (1995) was used to simulate a Dutch dairy farm before and after implementing the feeding strategies. The farm production plan was optimized based on the objective to maximize labor income, i.e. gross returns minus variable and fixed costs.

The LP model is a static year model and includes all relevant activities and constraints that are common to Dutch dairy farms, such as on-farm roughage production, purchase of feed, and animal production, including the rearing of young stock. The central element of the model is an average dairy cow from the Holstein Friesian breed, with a fixed annual milk production, calving in February, and representing the dairy cattle of the farm. The model distinguishes a summer period (183 days) and winter period (182 days) regarding feeding. Feed requirements (energy and protein) and intake capacity of the average cow were determined using the bio-economic model of Groen (1988). Safety margins for requirements of true protein digested in the small intestine and for rumen degradable protein balance were set at 100 g/cow per day. Based on feed restrictions, the LP model matches feed requirements of the cow with on-farm feed production and purchased feed.

On-farm feed production includes production of maize silage and production of grass for grazing and silage making. One hectare of maize silage yields 15.5 t DM per year, which equals 102 GJ

NE_L (CBS, 2013). Grassland yield depends on the level of N fertilization, which can vary from 100 to 500 kg/ha per year. Based on 225 kg N/ha per year, one hectare grassland yields 66 GJ NE_L/year. Purchased feeds include maize silage, three types of concentrates that differ in protein level (i.e. standard, medium and high) and dietary urea. All dietary options were available in winter and summer, except for fresh grass (only in summer). Table 1 shows the feed characteristics of the feeds that are standard available in the model; feed products that are available after implementing the strategies are discussed in the paragraph on feeding strategies. The prices of purchased feeds are shown in Table 2.

Constraints of the model include fixed resources of the farm (e.g. land area, milk quota, and housing capacity), links between activities (e.g. feed restrictions, link between manure production and application), and environmental policies. Cows are housed in a cubicle system with slatted floors and manure storage under the slats. Produced manure is applied with low emission techniques. The division of manure between grassland and maize land is optimized by the model. Environmental policies include limits to the application of total mineral N (in case of manure N,

Table 1. Feed characteristics of feeds standard available in the dairy farm model.

Feed product	NE_L[1] (MJ/kg DM)	DVE[2] (g/kg DM)	OEB[3] (g/kg DM)	N (g/kg DM)	Fill value[4] (kg/kg DM)	NDF (g/kg DM)	Crude fat (g/kg DM)
Concentrates							
- standard protein	7.21	100	6	24.1	0.29-0.72	414	48
- medium protein	7.21	133	28	32.2	0.29-0.72	407	51
- high protein	7.21	200	83	48.3	0.29-0.72	312	46
Dietary urea	0.00	0	2920	467.0	0.00	0	0
Fresh grass normal cut (1700 kg DM/ha)							
- 125 kg N	6.62	94	9	28.0	0.93	457	37
- 175 kg N	6.68	96	16	29.4	0.93	452	38
- 225 kg N	6.73	98	23	30.9	0.93	448	39
- 275 kg N	6.77	99	31	32.4	0.93	445	40
Grass silage normal cut (3500 kg DM/ha)							
- 125 kg N	5.89	70	22	25.6	1.08	506	35
- 175 kg N	5.93	71	31	27.4	1.08	501	36
- 225 kg N	5.97	73	39	29.0	1.08	497	37
- 275 kg N	6.00	74	47	30.6	1.08	493	39
Maize silage	6.56	58	-36	13.4	1.02	373	25

[1] Net energy for lactation.

[2] True protein digested in the small intestine according to Dutch standards (Tamminga et al., 1994).

[3] Rumen degradable protein balance according to Dutch standards (Tamminga et al., 1994).

[4] Fill value per kg DM feed expressed in kg of a standard reference feed (see Jarrige, 1988). The fill value of concentrates increases with an increase in concentrate intake.

Table 2. Costs and emission factors of purchased feeds.

	Price (€/t DM)	Emission factor (kg CO_2e/t DM)
Standard feeds		
Maize silage	148	182
Concentrates		
- standard protein	244	748
- medium protein	261	768
- high protein	322	801
Urea	528	1650
Feeds introduced with strategies		
Extruded linseed product	674[1]	1174
Nitrate source	1200[2]	727

[1] Based on the price of comparable high linseed products in the Netherlands.
[2] H.B. Perdok, Cargill Animal Nutrition, Velddriel, The Netherlands, personal communication.

2.2 kg N/ton is assumed organic N while the rest, depending on N in the diets is assumed to be mineral) and phosphate (P_2O_5) on the farm, and limits to the application of N from animal manure (DR Loket, 2012). The latter constraint is based on the European nitrate directive (Directive 91/676/EEC) and includes the derogation regulation, which is specific for a few countries in the EU that have a high proportion of grassland (EU, 2010). It prescribes that farms with at least 70% grassland can apply 250 kg N/ha originating from animal manure, instead of 170 kg N/ha that is prescribed for farms with less than 70% grassland. For a more detailed description of the model see Van Middelaar et al. (2013b).

2.2 Mechanistic model for enteric CH_4 production

Emission of enteric CH_4 from dairy cows was calculated using a mechanistic model originating from Dijkstra et al. (1992), and modified and updated by Mills et al. (2001) and applying volatile fatty acid (VFA) stoichiometry of Bannink et al. (2006). The model simulates digestion, absorption, and outflow of nutrients in the rumen, small intestine and hindgut, and includes interactions between feed substrates and ruminal microbes, and production of volatile fatty acids, microbial mass, and di-hydrogen (H_2). Production of CH_4 is estimated from the H_2 balance. Sources of H_2 include H_2 from production of acetate and butyrate, and from microbial growth with amino acid as N-source. Sinks of H_2 include H_2 used for production of propionate and valerate, for microbial growth with ammonia as N-source, and for bio-hydrogenation of lipids. The surplus H_2 is assumed to be completely converted into CH_4.

To calculate the effect of dietary supplementation of extruded linseed and nitrate on enteric CH_4 production, additional calculations were required. These calculations are described in the paragraph on feeding strategies.

2.3 Life cycle assessment

Life cycle assessment, an internationally accepted and standardized method (ISO 14040 and ISO 14044), was used to evaluate GHG emissions (including CO_2, CH_4, and N_2O) along the milk production chain, up to the moment that milk leaves the farm gate. Processes included are the extraction of raw materials to produce farm inputs, the manufacturing and distribution of these inputs, and all processes on the dairy farm. Stages related to transport and processing of milk were assumed to be unaffected by the strategies, and, therefore, not included in the analysis. Different GHG emissions were summed up based on their equivalence factor in terms of CO_2 equivalents (CO_2e) (100-year time horizon): 1 for CO_2, 25 for CH_4, and 298 for N_2O (Forster et al., 2007). After summing up emissions they were allocated to the different outputs of the farm (i.e. milk and meat) based on the relative economic value of these outputs (i.e. economic allocation): 89% of the emissions were allocated to milk and 11% to meat. Economic allocation is used frequently in LCA studies of livestock products (De Vries and De Boer, 2010). Emissions were divided by the total amount of FPCM and expressed in kg CO_2e/ton FPCM.

Emissions from the production of farm inputs were based on Eco-invent (2007) (synthetic fertilizer, pesticides, tap water, energy sources); on Vellinga et al. (2013) (concentrates, dietary urea, milk replacer); and on own calculations (purchased maize silage). For a detailed description of the calculations see Van Middelaar et al. (2013b). Final CO_2e per ton DM of purchased feeds are included in Table 2.

Emission of enteric CH_4 from young stock was based on IPCC Tier 2 methods and default values (IPCC, 2006). Emissions of CH_4 from manure management were based on national inventory reports, i.e. 0.746 kg CH_4 per ton manure produced in stables, and 0.110 kg CH_4 per ton manure produced during grazing (De Mol and Hilhorst, 2003). Emissions of CO_2 from the combustion of diesel and gas during on-farm processes were based on Eco-invent (2007). Direct and indirect N_2O emissions (the latter resulting from volatilization of NH_3 and NO_x and from leaching of NO_3^-) from manure management and from N application to the field (including N from manure, synthetic fertilizers, and crop residues) were based on national inventory reports and IPCC (2006). For a detailed description of the calculations see Van Middelaar et al. (2013b).

2.4 Feeding strategies

Extruded linseed (LINS)

Extruded linseed was added as a commercially available linseed product described by Dang Van et al. (2008), containing 56.0% crushed linseed, 21.0% wheat, 15.0% sunflower cake, 4.5% field beans, 2.0% butylated hydroxytoluene, 1.0% linseed oil, and 0.5% salt. Table 3 shows feed characteristics of this product, and Table 2 shows prices. Considering that high amounts of dietary fat can have negative effects on DMI, digestibility, and milk production (Grainger and Beauchemin, 2011), 1 kg product/cow per day was prescribed in the diet in summer and 2 kg/cow per day in winter (the product contains 0.9 kg DM/kg product). Addition of dietary fat in the form of extruded linseed reduces enteric CH_4 production because unsaturated fatty acids provide an alternative H_2 sink and prevent the formation of CH_4 from CO_2 and H_2. In addition, adding fat may primarily inhibit fibrolytic bacteria and cause a shift in volatile fatty acid production towards propionate, reducing CH_4 production (Ellis et al., 2008).

Table 3. Feed characteristics of feeds available after implementing the strategies.

Feed product	NE_L[1] (MJ/kg DM)	DVE[2] (g/kg DM)	OEB[3] (g/kg DM)	N (g/kg DM)	Fill value[4] (kg/kg DM)	NDF (g/kg DM)	Crude fat (g/kg DM)
LINS							
Extruded linseed prod.	10.51	96	87	36.9	0.29	209	236
NITR							
Nitrate source	0.00	0	1170	187.3	0.00	0	0
GMS [5]							
Fresh grass early cut (1400 kg DM/ha)							
- 125 kg N	6.67	96	10	28.9	0.93	442	37
- 175 kg N	6.72	98	18	30.4	0.93	437	39
- 225 kg N	6.77	100	26	31.9	0.93	433	40
- 275 kg N	6.82	102	35	33.5	0.93	430	41
Grass silage early cut (3000 kg DM/ha)							
- 125 kg N	5.96	73	27	27.6	1.08	488	36
- 175 kg N	6.01	74	38	29.5	1.08	484	38
- 225 kg N	6.04	76	48	31.3	1.08	480	39
- 275 kg N	6.08	77	58	33.0	1.08	476	41

[1] Net energy for lactation.
[2] True protein digested in the small intestine according to Dutch standards (Tamminga et al., 1994).
[3] Rumen degradable protein balance according to Dutch standards (Tamminga et al., 1994).
[4] Fill value per kg feed expressed in kg of a standard reference feed (see Jarrige, 1988).
[5] Feed characteristics of grass and grass silage were based on national reports that describe characteristics of grass and grass silage at different dry matter yields (CVB, 2011). Missing values were interpolated assuming a linear relation between two values.

The effect of adding fatty acids in the form of extruded linseed on CH_4 production was based on Grainger and Beauchemin (2011). They performed a meta-analysis using data from 27 studies to determine the effect of dietary fat on CH_4 production. The reduction in CH_4 production was calculated by the following equation:

$y = -0.102\,x$

where, y is the reduction in enteric CH_4 (g/kg DM intake); and x is the total amount of dietary fat added (g/kg DM).

Emissions related to the production of the extruded linseed product were based on Vellinga et al. (2013). The method was similar to the one that was used to calculate the impact of concentrates production. Table 2 shows final CO_2e per ton DM.

Nitrate (NITR)

A nitrate source ($5Ca(NO_3)_2 \cdot NH_4NO_3 \cdot 10H_2O$; 75% NO_3 in DM) was added at 1% of dietary DM. Table 3 shows feed characteristics of this nitrate source, and Table 2 shows prices. In the rumen, nitrate is reduced to nitrite and, subsequently, nitrite is reduced to ammonia. These processes provide an alternative H_2 sink that is energetically more favorable than reduction of CO_2 to CH_4 (Ungerfeld and Kohn, 2006).

The effect of dietary nitrate on CH_4 production was based on Van Zijderveld (2011). Stoichiometrically, a reduction in CH_4 of 0.258 g/g nitrate is expected. *In vivo*, efficiency of CH_4 reduction decreases with increased levels of nitrate intake according to the following equation:

$y = -0.17\,x + 1.13$

where, y is the actual reduction in enteric CH_4 expressed as a fraction of the reduction potential according to stoichiometry; and x is the amount of nitrate expressed in g/kg metabolic weight ($kg^{0.75}$) per day. The body weight of the cow is assumed to be 650 kg, which equals a metabolic weight of 129 kg.

Emissions related to the production of dietary nitrate were based on Eco-invent (2007). Table 2 shows final CO_2e per ton DM.

Grass maturity stage (GMS)

Reducing the maturity stage of grass and grass silage results in a lower DM yield/ha per year, but increases grass quality in terms of energy and protein content per kg DM. Total yield in MJ

NE_L/ha per year was assumed to remain unchanged. In the reference situation, grazing was applied at 1700 kg DM/ha, and harvesting at 3500 kg DM/ha (Berentsen and Giesen, 1995). After implementing the strategy, grazing was applied at 1400 kg DM/ha, and harvesting at 3000 kg DM/ha. Table 3 shows feed characteristics of less mature grass and grass silage. These less mature grass products have a lower NDF and a higher protein and fat content compared with grass products in the reference situation (based on CVB, 2011). Feeding less mature grass, therefore, may shift the profile of volatile fatty acids in the rumen towards higher propionic acid levels, and consequently reduces the production of enteric CH_4 per unit feed NE or milk. Less mature grass products, moreover, have a higher digestibility. Assuming a constant milk production, a higher digestibility in combination with the higher nutritional value will reduce total DM intake, and as a result lower enteric CH_4 production. Costs per grass cut were assumed to be the same as in the reference situation. Due to a lower DM yield per grass cut, the number of cuts per year increases.

2.5 Set up of the analysis

We evaluated the cost-effectiveness of the three strategies for an average Dutch dairy farm on sandy soil. This average farm has 44.9 ha of land, housing facility for 76 dairy cows with young stock, and a milk quota of 603 tons per year. Milk production per cow was assumed to be constant at 7968 kg/year (4.39% fat and 3.52% protein). Data were based on the Farm Accountancy Data Network (FADN) of the Agricultural Economics Research Institute from the Netherlands (FADN, 2012).

Two additional feeding constraints were considered for all situations: 1) the maximum fresh grass intake in summer was assumed 12 kg DM/cow per day, because limited grazing was applied (Taweel et al., 2004; Abrahamse et al., 2009). Limited grazing (i.e. grazing during daytimes) is the most common grazing regime on Dutch dairy farms (FADN, 2012); 2) the maximum amount of NPN in the diet was assumed equal to the amount of NPN in the diet supplemented with nitrate.

The reference situation, which includes no predefined feeding strategy, was determined by maximizing labor income for this average Dutch dairy farm. Subsequently, one of the three feeding strategies was introduced. Labor income of the farm was maximized again to determine diets and farm plan after implementing each strategy.

Cost-effectiveness

The cost-effectiveness of the feeding strategies represents the costs per unit of GHG reduction. It is calculated by dividing the decrease in labor income of the farm family (€/year) by the decrease in GHG emissions at chain level (kg CO_2e/year). To account for the total reduction in GHG emissions, all emissions at farm were considered when calculating the cost-effectiveness, implying that no economic allocation was used.

Uncertainty in prices and emission factors

Prices and emission factors contain uncertainty which can influence results (Van Middelaar et al., 2013a). To quantify the impact of this uncertainty, an uncertainty analysis was performed. Prices of feed products (i.e. purchased maize silage, concentrates, urea, extruded linseed product, and nitrate) were changed by ± 25%, equal to the variation in prices of concentrates observed over the last 10 years (KWIN, 2001-2013). Emissions related to production of these products were also changed by ± 25%, equal to the variation in emissions of concentrate ingredients found in literature (Nguyen et al., 2012; Van Middelaar et al., 2013a; Vellinga et al., 2013). Uncertainty related to calculating enteric CH4 production varied between strategies. In the case of LINS, uncertainty was based on Grainger and Beauchemin (2011) and equaled ± 14.4% of the calculated reduction resulting from dietary supplementation of fatty acids. In the case of NITR, uncertainty was based on Van Zijderveld et al. (2011) and equaled ± 14.2% of the calculated reduction resulting from adding nitrate. In the case of GMS, uncertainty was based on Bannink et al. (2011) and equaled ± 13.0% of the total enteric CH_4 production/t FPCM in the reference situation and after reducing grass maturity. All prices were increased or decreased by 25% at the same time, because price fluctuations of different products were assumed to be related. Emission factors were changed independently, because they were assumed to be unrelated.

3 Results and discussion

3.1 Diets, farm plan, and labor income

In all situations the milk quota of 603 tons was fully used, resulting in a dairy herd consisting of 75.7 cows and 24.9 young stock units (i.e. one unit includes 1 animal <12 months and 0.96 animal >12 months). The diets of young stock consisted of milk replacer and concentrates in the first months followed by grass and concentrates during the summer period and maize silage and concentrates during the winter period.

Table 4 shows the diets of the dairy cows and farm plan for the reference situation and the situations after implementing the feeding strategies. In the reference situation, the maximum amount of fresh grass is fed in summer, because this is the cheapest way of feeding. Maize silage is added up to 6.59 kg DM/cow per day in combination with standard protein concentrates and dietary urea. As a result, minimum requirements for energy and rumen degradable protein are met within the limiting intake capacity. In winter, 2.86 kg DM grass silage/cow per day is fed, which is the amount of grass left for ensiling after grazing, in combination with 10.98 kg DM maize silage/cow per day. High protein concentrates and urea were added to meet requirements for energy, rumen degradable protein, and true protein digested in the small intestine. Seventy per cent of the farm land was used as grassland and thirty per cent as maize land. Labor income of the farm family was €42,605 per year. This matches with the average income of a farm family in practice, which was €42,900 in 2010 (dairy farm on sandy soil; FADN, 2012).

Feeding strategy LINS increased the fat content of the summer diet from 35 g/kg DM (reference situation) to 44 g/kg DM. As a result, total DM intake reduced. The amount of maize silage decreased by 0.52 kg DM/cow per day, standard concentrates and urea were removed from the diet, and 0.04 kg DM/cow per day of high protein concentrates was added. In winter, dietary fat content increased from 32 g/kg DM in the reference situation to 56 g/kg DM after implementing LINS. As a result, the amount of maize silage decreased by almost 3 kg DM/cow per day, and urea was removed from the diet. The amount of high protein concentrates remained to fulfill requirements for true protein digested in the small intestine. Due to the dietary changes, the amount of purchased maize silage and concentrates decreased. Labor income reduced to €26,564 per year. This reduction is caused almost completely by the relatively high costs of the extruded linseed product compared to the costs of maize silage and concentrates.

Feeding strategy NITR resulted in a dietary NPN level of 37 g/cow per day in summer, and 31 g/cow per day in winter, being the maximum amount of dietary NPN allowed. As a result, urea was removed from the diet. No other dietary changes occurred. Due to an increase in dietary N content, the amount of N in manure increased. As a result, the amount of synthetic fertilizer decreased. No other changes in farm production plan occurred. When application standards for animal manure would be restricting, feeding nitrate could result in a situation where manure has to be removed from the farm, or additional dietary changes would be required to reduce the amount of N in manure. This was not the case in the present study. Labor income reduced to €37,142 per year. This reduction is caused by the higher costs of dietary nitrate compared with urea.

Table 4. Diets and farm plan for the reference situation and after implementing one of the three strategies[1].

		REF	LINS	NITR	GMS
Diet dairy cows - summer period (kg DM/cow/day)					
Grass (grazed)		12.00	12.00	12.00	12.00
Maize silage		6.59	6.07	6.59	6.62
Concentrates - standard protein		0.88	-	0.88	0.78
- high protein		-	0.04	-	-
Urea		0.02	-	-	0.01
Extruded linseed product		-	0.90	-	-
Nitrate source		-	-	0.20	-
Diet is restricted by [2]		E,I,R	E,T	E,I	E,I,R
Diet dairy cows - winter period (kg DM/cow/day)					
Grass silage		2.86	2.86	2.86	2.75
Maize silage		10.98	8.14	10.98	11.09
Concentrates - high protein		2.40	2.36	2.40	2.37
Urea		0.06	-	-	0.06
Extruded linseed product		-	1.80	-	-
Nitrate source		-	-	0.16	-
Diet is restricted by [2]		E,R,T	E,T	E,T	E,R,T
On-farm feed production					
Grassland 225 kg N	ha	31.4	31.4	31.4	31.4
Maize land	ha	13.5	13.5	13.5	13.5
Farm inputs					
Synthetic fertilizer	kg N/ha	117	118	111	116
	kg P_2O_5/ha	8	7	7	10
Maize silage	t DM	96	48	96	98
Concentrates	t DM	55	43	55	53
Urea	t DM	1	-	-	1
Extr. linseed product	t DM	-	38	-	-
Nitrate source	t DM	-	-	5	-
Labor income [3]	€	42,605	26,564	37,142	42,142

[1] LINS = feeding an extruded linseed product (1 kg/cow per day in summer and 2 kg/cow per day in winter)
NITR = feeding a nitrate source (1% of DM)
GMS = reducing maturity stage of grass and grass silage

[2] The diet can be restricted by: E = energy requirements; R = rumen degradable protein balance; T = true protein digested in the small intestine; I = intake capacity.

[3] Milk was sold at a price of €310 per ton (Wageningen UR, 2011). Culled cows were sold for €525 per cow. Agricultural subsidy was included and added up to about €32,500/year.

Feeding strategy GMS did not affect the amount of grass in kg DM/cow per day in the summer diet. Due to a higher energy content and a higher rumen degradable protein content per kg grass, however, the amount of concentrates and urea slightly decreased and that of maize silage slightly increased. Because total DM yield per ha grassland decreased, the amount of grass silage in the winter diet decreased. Maize silage slightly increased, while the amount of concentrates and urea remained unchanged. Due to a higher N and a lower P content in the diet, the amount of N in manure increased, while the amount of P decreased. This is reflected by a change in purchased fertilizers. Labor income reduced to €42.142 per year. This reduction is caused mainly by an increase in costs related to grassland management, resulting from an increase in the number of grass cuts per ha per year. In addition, costs of purchased maize silage increased, while costs of purchased concentrates decreased.

3.2 Greenhouse gas emissions

Table 5 shows GHG emissions in the reference situation and the effect of implementing the feeding strategies. Emissions in the reference situation added up to 840 kg CO_2e/t FPCM. The most important contributor is enteric CH_4 (52%), followed by emissions from manure (14%), on-farm feed production (13%), purchased feed products (10%), and synthetic fertilizers (8%).

Emissions per t FPCM are low compared with results in literature (De Vries and De Boer, 2010; Flysjö et al., 2011a; Zehetmeier et al., 2012). The lower emissions per t FPCM have three main causes. First, the diets contain relatively high amounts of maize silage and low amounts of concentrates, partly because urea was used. Compared to concentrates, maize silage results in less emissions during production (see Table 2), and in less enteric CH_4 production. Second, we used a mechanistic model to calculate enteric CH_4 production, whereas most other studies use IPCC Tier 2 methods that generally overestimate enteric CH_4 (Kebreab et al., 2008; Alemu et al., 2011). Third, unlike most other studies we used a model farm and calculated feed intake, which may differ from the actual intake and may increase the efficiency of the farm.

LINS reduced emissions of enteric CH_4 from dairy cows by 42 kg CO_2e/t FPCM. Due to a decrease in the amount of purchased maize silage, concentrates, and urea, emissions related to the production of these products decreased by 29 kg CO_2e/t FPCM in total. Emissions from the production of the extruded linseed product added up to 63 kg CO_2e/t FPCM. Changes in other emissions were minor and relate to an increase in the P content of manure. Overall, emissions reduced by 9 kg CO_2e/t FPCM.

Table 5. Greenhouse gas emissions for the reference situation and the effect on emissions of implementing the feeding strategies[1] (kg CO_2e/t FPCM[2]; based on an economic allocation factor of 89%).

	REF	LINS	NITR	GMS
Animal emissions				
Enteric CH_4 emission dairy cows [3]	360	-42	-33	-10
Enteric CH_4 emission young stock	79	0	0	0
Manure [4]	114	0	3	1
On-farm feed production				
Grassland [5]	70	0	-1	0
Maize land [6]	40	-2	-1	-1
Production of farm inputs				
Maize silage	25	-13	0	1
Concentrates	60	-13	0	-2
Urea	3	-3	-3	0
Composed linseed product	-	63	-	-
Nitrate source	-	-	5	-
Synthetic fertilizer	65	1	-3	0
Other inputs [7]	25	0	0	0
Total emissions	840	-9	-32	-11

[1] LINS = feeding an extruded linseed product (1 kg/cow per day in summer and 2 kg/cow per day in winter)
NITR = feeding a nitrate source (1% of DM)
GMS = reducing maturity stage of grass and grass silage
[2] Different GHG emissions were summed up based on their equivalence factor in terms of CO_2 equivalents: 1 for CO_2, 25 for CH_4, and 298 for N_2O (Forster et al., 2007).
[3] Enteric CH_4 production in g CH_4/cow per day was: REF 428 (summer), 323 (winter); LINS 404 (summer), 260 (winter); NITR 390 (summer), 292 (winter); GMS 418 (summer), 312 (winter).
[4] Including emissions from grazing (about 55%, of which about 97% N_2O and 3% CH_4) and from manure storage (about 45%, of which about 12% N_2O and 88% CH_4).
[5] Including N_2O emissions from N application (about 92%) and emissions related to combustion of diesel during field work (about 8%).
[6] Including N_2O emissions from N application (about 80%) and emissions related to combustion of diesel during field work (about 20%).
[7] Including milk replacer, bedding material, energy sources, tap water, and machinery for field work.

NITR reduced emission of enteric CH_4 from dairy cows by 33 kg CO_2e/t FPCM. Producing nitrate instead of urea increased emissions by 3 kg CO_2e/t FPCM. Changes in other emissions are minor and relate to an increase in the N content of manure. Overall, emission reduced by 32 kg CO_2e/t FPCM.

GMS reduced emissions of enteric CH_4 from dairy cows by 10 kg CO_2e/t FPCM. Changes in other emissions were minor and relate to changes in the diet and an increase in the N content of manure. Overall, emissions reduced by 11 kg CO_2e/t FPCM.

3.3 Cost-effectiveness

The cost-effectiveness of the strategies is €2594/t CO_2e for LINS; €241/t CO_2e for NITR; and €57/t CO_2e for GMS. The allowance price of CO_2 (i.e. 'the market value') has been €30/t CO_2 at its maximum since the introduction of the EU Emissions Trading System in 2005 (Calel, 2013). Compared to this value, the cost-effectiveness of all three strategies is very low. No other studies were found that calculated the cost-effectiveness of feeding strategies based on economic optimization.

Figure 1 shows the results of the uncertainty analysis. When we only consider the reduction in labor income, i.e. the costs of the strategy (y-axis), we see that GMS has the lowest costs, then NITR, and then LINS. The uncertainty ranges do not overlap, showing that it is unlikely that a change in price factors will affect the order of the strategies. When we only consider the impact on emissions (x-axis), we see that the reduction was largest for NITR, then for GMS, and then for LINS. The uncertainty range of LINS, however, is very large. A change in emission factors (e.g. by

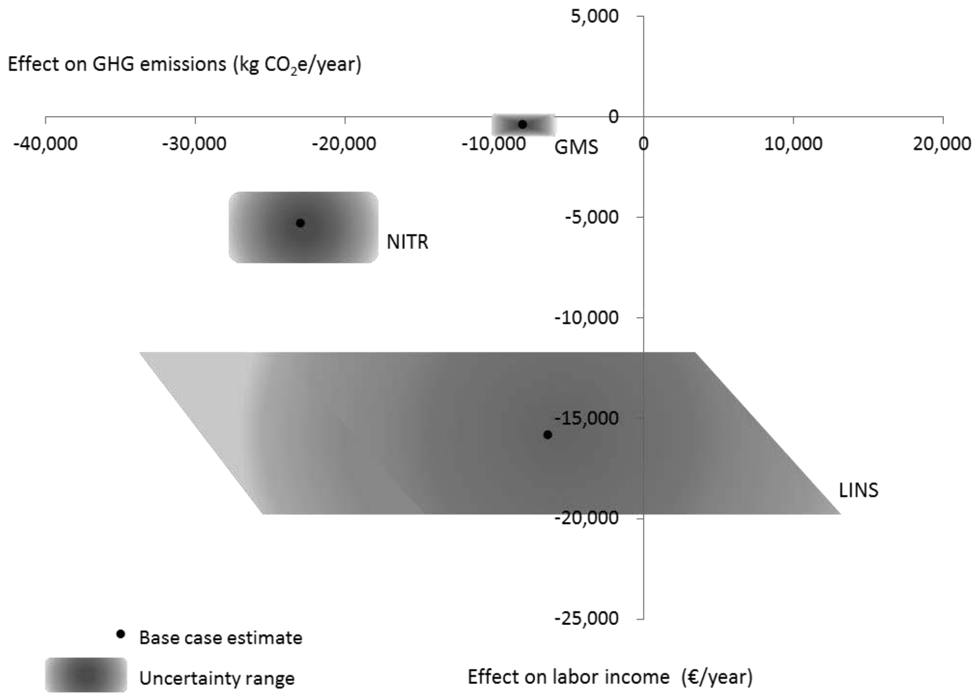

Figure 1. Cost-effectiveness of the feeding strategies: feeding an extruded linseed product (LINS); feeding a dietary nitrate source (NITR); and reducing the maturity stage of grass and grass silage (GMS).

Table 6. Cost-effectiveness of the feeding strategies[1] (€/t CO_2 reduced).

	Cost-effectiveness	Uncertainty range
LINS	2594	[349 ; **]
NITR	241	[149 ; 381]
GMS	57	[40 ; 86]

[1] LINS = feeding an extruded linseed product
NITR = feeding a nitrate source
GMS = reducing maturity stage of grass and grass silage
** Emissions increased and therefore the upper limit of the uncertainty range cannot be calculated.

creating production circumstances that result in lower emissions), therefore, can increase the relative importance of LINS compared to GMS and NITR. LINS has a large uncertainty range because not only enteric CH_4 production was affected, but also emissions from production of purchased feed products. In case of NITR and GMS, changes in emissions other than enteric CH_4 production are less important.

The uncertainty range of LINS has the shape of a parallelogram, whereas the uncertainty range of NITR and GMS have the shape of a rectangle. The parallelogram arises because for LINS decreasing the price of purchased feed products also changed the farm plan. Changes in GHG emissions, therefore, not only resulted from a change in underlying emission factors, but also from a change in farm plan. Because the change in farm plan resulted in lower emissions, decreasing the prices of purchased feed products resulted in an extra emission reduction, creating the shape of a parallelogram. Most important changes include a reduction in the amount of N application on grassland from 225 kg/ha to 200 kg/ha, a reduction in the amount of synthetic fertilizers (from 118 to 102 kg N/ha), and an increase in the amount of purchased maize silage (from 48 to 55 t DM) and concentrates (from 43 to 45 t DM) to compensate for a decrease in DM yield per ha grassland, and a decrease in nutritional value of grass and grass silage.

Table 6 shows the cost-effectiveness of the strategies including the uncertainty in price and emission factors. Results show that GMS is most cost-effective, then NITR, and then LINS, and that changes in prices and emission factors are unlikely to change this order.

4 General discussion

Supplementation of an extruded linseed product resulted in a dietary fat content of 44 g/kg DM in summer and 56 g/kg DM in winter. As a result, enteric CH_4 reduced by 1.9 g/kg FPCM. Martin et al. (2008) found a reduction of 4.5 g/kg FCM when adding extruded linseed to achieve a dietary fat content of 57 g/kg DM. The reduction found by Martin et al. (2008), however, mainly originated from a reduction in feed intake and feed digestibility. In addition, milk production decreased significantly in the study of Martin et al. (2008).

The reduction in labor income in case of LINS was very large, resulting in a cost-effectiveness of only €2594/t CO_2e. Because the diets in the reference situation contain relatively high amounts of maize silage and low amounts of concentrates, not only concentrates, but also maize silage was replaced by the extruded linseed product. Based on energy content, maize silage is cheaper than concentrates and extruded linseed. The reduction in labor income, therefore, is larger than when the extruded linseed product would solely replace concentrates.

Reducing the maturity stage of grass and grass silage reduced enteric CH_4 production by 0.5 g/kg FPCM. Brask et al. (2013) found a reduction of about 2.2 g CH_4/kg ECM, when comparing the impact of feeding grass silage from an early cut with grass silage from a late cut. Differences in quality (e.g. NE_L and protein content per kg DM) between the two silages, however, were much larger than in our study. Because our reference situation was based on a diet using grass products from a normal cut, increasing grass quality by reducing the maturity stage of grass is limited. This will probably also be the case for most farms in practice.

Including high amounts of fat (i.e. > 70 g/kg DM) can negatively affect DMI and fiber digestion in the rumen (Schroeder et al., 2004; Grainger and Beauchemin, 2011). This can reduce milk yield, and affect milk composition. Based on the amounts provided in the present study (i.e. 44 g/kg DM in summer and 56 g/kg DM in winter), negative effects are not expected. To maximize the reduction in GHG emissions, the amount of extruded linseed could be further increased, i.e. up to a dietary fat content of 70 g/kg DM. An increased use of linseed, however, will further decrease labor income. Other fat sources, such as canola oil and cottonseeds, might provide an alternative with better cost-effectiveness.

When energy intake is the limiting factor for milk production, fat supplementation or reducing the maturity stage of grass can increase milk yield per cow (Schroeder et al., 2004; Weiss and Pinos-Rodríguez, 2009). In addition, reducing the maturity stage of grass can increase DMI (Brask et al., 2013), which can also increase milk yield per cow. In our study, milk yield per cow was kept constant, and diets were adjusted so that energy requirements were met before and after implementing the strategies. Diets were restricted not only by energy requirements, but also by requirements for rumen degradable protein balance, intake capacity, and requirements for true protein digested in the small intestine (Table 4). In situations where nutrients other than energy are limiting, fat supplementation is unlikely to increase milk yield. Generally, this will be the case in high forage diets, whereas in low forage diets fat supplementation potentially increases milk yield (Weiss and Pinos-Rodríguez, 2009). Reducing grass maturity offers potential to increase milk yield in situations where not only energy, but also other nutrients and intake capacity are limiting. An increase in milk yield, provided that health and fertility parameters do not decrease, will improve the cost-effectiveness of feeding strategies to reduce GHG emissions (Van Middelaar et al., 2014)

Assuming typical human consumption levels in various western countries, milk fat contributes up to 34% of the daily intake of various long-chain omega-3 and omega-6 FA (Van Valenberg et al., 2013). This indicates that a substantial part of the intake of those fatty acids by humans come from milk fat. Addition of unsaturated FA to a dairy cow's diet can change the FA profile of milk towards less saturated medium-chain FA and more long-chain unsaturated FA (Sterk et al., 2012). This change in milk FA profile is considered to be good for human health (Kliem and Givens, 2011), and offers an opportunity to increase the revenues per kg milk if consumers are willing to pay a higher price for milk with enhanced proportions of unsaturated fatty acids. In our study, we did not account for this effect, which can occur from LINS and to a lower extent from GMS (Glasser et al., 2008). A positive effect on milk FA profile and the consequential increase in revenues can improve the cost-effectiveness of LINS and GMS.

A dietary nitrate source was added at 1% of DM. Feeding high levels of nitrate to animals that are not yet adapted can cause methemoglobinemia, a blood disorder in which haemoglobin is unable to release oxygen to the body tissue. Methemoglobinemia can be caused by increased levels of nitrite in the rumen and subsequent absorption due to lack of nitrite reducing bacterial activity. In a previous study, the same nitrate source was fed to dairy cows at 2.1% of DM without any negative consequences for animal health. These cows went through an adaptation period of three weeks, with weekly increments of 25% of the final level (Van Zijderveld et al., 2011). Similarly, no negative consequences for animal health were observed in sheep that were fed nitrate at 2.6% of DM after a similar adaptation regime of three weeks (Van Zijderveld et al., 2010), and in steers that were fed nitrate at 2.2% of DM after an adaptation period of 12 days, with 4 days increments of 25% of the final level (Hulshof et al., 2012). Feeding nitrate at 1% of DM, therefore, is assumed to pose no risk for methemoglobinemia.

An important source of GHG emissions related to agriculture is land use change (LUC). In this study, we did not include LUC emissions because the strategies did not affect on-farm land use, and the impact on type and amount of purchased feed ingredients related to LUC was limited (Van Middelaar et al., 2013a).

We evaluated the effect of the strategies for an average Dutch dairy farm on sandy soil. In summer, the main forage type was fresh grass from grazing, in winter it was maize silage. For farms with a similar diet, within and outside the Netherlands, result can be used as an indicator to estimate the impact of the strategies on GHG emissions. For farms with a different diet, further analysis is required.

In 2015 the EU milk quota system will be abolished. This might change the Dutch dairy system (Louhichi et al., 2010). The conclusions of this paper, however, are assumed to stay valid after abolition of the milk quota system. Intensification of Dutch dairy farms will be limited by environmental policies such as the European nitrate directive (EL&I, 2009), and given the growing conditions for crop production in the Netherlands, grass is expected to remain the favorite type of forage and major changes in the diet are not expected.

All three strategies resulted in a reduction in labor income. This reduces the likelihood of adoption by farmers, because profitability is often the main driver in decision making (Hristov et al., 2013b). Because GMS resulted in the lowest additional costs and the best cost-effectiveness, this strategy seems to be most promising for application in practice, especially in case future legislation or subsidies might provide extra stimuli to implement mitigation options.

5 Conclusions

We evaluated the cost-effectiveness of three feeding strategies to reduce enteric CH_4 production in dairy cows, by calculating the impact on profitability at farm level and on GHG emissions at chain level. Reducing the maturity stage of grass and grass silage was most cost-effective (€57/t CO_2e), then supplementation of dietary nitrate (€241/t CO_2e), and then supplementation of an extruded linseed product (€2594/t CO_2e). Supplementation of nitrate resulted in the largest reduction in GHG emissions, but reducing the maturity stage of grass and grass silage resulted in lower costs, and a better cost-effectiveness. This latter strategy, therefore, was found to be most promising for application in practice.

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Chapter 5

Methods to determine the relative value of genetic traits in dairy cows to reduce greenhouse gas emissions along the chain

C.E. van Middelaar[a], P.B.M. Berentsen[b], J. Dijkstra[c], J.A.M. van Arendonk[d], and I.J.M. de Boer[a]

[a] Animal Production Systems group, Wageningen University, Wageningen, the Netherlands

[b] Business Economics group, Wageningen University, Wageningen, the Netherlands

[c] Animal Nutrition group, Wageningen University, Wageningen, the Netherlands

[d] Animal Breeding and Genomics Centre, Wageningen University, Wageningen, the Netherlands

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Abstract

Current decisions on breeding in dairy farming are mainly based on economic values of heritable traits, since earning an income is a primary objective of farmers. Recent literature, however, shows that breeding also has potential to reduce greenhouse gas (GHG) emissions. The objective of this paper was to compare two methods to determine GHG values of genetic traits. Method 1 calculates GHG values using the current strategy (i.e. maximizing labor income), whereas method 2 is based on minimizing GHGs per kg milk and shows what can be achieved if the breeding results are fully directed at minimizing GHG emissions. A whole-farm optimization model was used to determine results before and after one genetic standard deviation improvement (i.e. unit change) of milk yield and longevity. The objective function of the model differed between method 1 and 2. Method 1 maximizes labor income. Method 2 minimizes GHG emissions per kg milk while maintaining labor income and total milk production at least at the level before the change in trait. Results show that the full potential of the traits to reduce GHG emissions given the boundaries that were set for income and milk production (453 kg CO_2equivalents (CO_2e)/unit change per cow per year for milk yield and 441 for longevity) is about twice as high as the reduction based on maximizing labor income (247 kg CO_2e/unit change per cow per year for milk yield and 210 for longevity). The GHG value of milk yield is higher than that of longevity, especially when the focus is on maximizing labor income. Based on a sensitivity analysis it was shown that including emissions from land use change and using different methods for handling the interaction between milk and meat production can change results, generally in favor of milk yield. Results can be used by breeding organizations that want to include GHG values in their breeding goal. To verify GHG values, the effect of prices and emissions factors should be considered, as well as the potential effect of variation between farm types.

1 Introduction

The need for strategies to reduce greenhouse gas (GHG) emissions from human activities, mainly consisting of carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O), has been highlighted (IPCC, 2007). Use of fossil fuel and land use change are identified as the primary sources for increased levels of atmospheric CO_2, whereas agriculture is identified as the primary source for increased levels of CH_4 and N_2O (IPCC, 2007). The majority of CH_4 emissions from agriculture relate to enteric fermentation of ruminants. About half of the total GHG emissions along the dairy production chain is enteric CH_4 (Hörtenhuber et al., 2010). In order to reduce CH_4 emissions, different strategies have been proposed. One of these strategies is increasing the productivity and efficiency of the dairy herd by selective breeding (De Haas et al., 2011; Buddle et al., 2011).

Productivity and efficiency can be increased by genetic improvement of traits such as milk yield, feed efficiency, longevity, and calving interval (Bell et al., 2011). Increasing milk yield per cow, for example, reduces CH_4 emissions per kg of milk by diluting CH_4 formed during fermentation of feed related to maintenance (Bell et al., 2010; Bell et al., 2011; Bannink et al., 2011). Bannink et al. (2011) showed that a 33% increase in production of fat-and-protein-corrected milk (FPCM), from 17.2 kg/d in 1990 to 22.9 kg/d in 2008, reduced enteric CH_4 per kg FPCM by 13%, from 17.6 to 15.4 g. Increasing longevity reduces CH_4 per kg of milk by reducing the number of female replacements producing CH_4 for maintenance and growth, without producing milk (Garnsworthy, 2004; Wall et al., 2010). Wall et al. (2010) showed that increasing longevity from an average of 3.0 to 3.5 lactations can reduce enteric CH_4 per kg milk by 4.4%.

Changing a trait, such as milk yield or longevity, however, can affect the whole farm, including feeding strategy, management practices and purchases of inputs like concentrate and fertilizer (Wall et al., 2010; Bell et al., 2010). Evaluating the impact of a genetic improvement, therefore, requires modeling the whole farm. Moreover, optimization of farm management before and after a change in trait is required to prevent under- or overestimation of the impact of genetic improvement (Groen et al., 1997). Finally, if the impact concerns GHG emissions, the analysis should include emission along the chain, i.e. from production of farm inputs up to the farm gate, to avoid pollution swapping. By evaluating the impact of one unit change in individual traits on GHG emissions at chain level, the relative value of each trait to reduce GHG emissions along the chain can be determined. A similar approach is used to calculate the relative economic value of traits (Groen, 1988; Koenen et al., 2000).

Two studies evaluated the impact of improving individual traits in dairy cows on GHG emissions at farm or chain level. Wall et al. (2010) evaluated the impact of increasing longevity on CH_4 and N_2O emissions at farm level, whereas Bell et al. (2011) evaluated the impact of increasing feed efficiency, milk yield, calving interval and longevity on GHG emissions at chain level. Both Wall et al. (2010) and Bell et al. (2011), however, did not optimize farm management with changing levels of genetic traits.

Farm management can be optimized based on different objectives like maximizing labor income (i.e. the main interest in deriving breeding objectives), or minimizing GHG emissions per unit product. It is not clear how a difference in objective affects the relative value of individual traits to reduce GHG emissions per kg FPCM.

The objective of this study was to compare two methods to determine the relative value of genetic traits in dairy cows to reduce GHG emissions along the milk production chain (i.e. up to the farm gate). Both methods are based on a whole-farm dairy model, use linear programming (LP) to optimize farm management, and include all GHG emissions along the chain up to the farm gate. The first method is based on maximizing labor income of the farm family; the interrelated consequences for GHG emissions are evaluated as a side-effect. The second method is based on minimizing GHG emissions per kg milk. We compared both methods by assessing the consequences of an increase in milk yield and longevity of cows on an average Dutch dairy farm on sandy soil.

2 Methods

The first method is based on the exact same principle that is used to calculate economic values. A dairy farm LP model with the objective to maximize labor income was used to determine the economic benefit per unit change in milk yield and longevity. The effect on GHG emissions (i.e. the GHG value) was considered as a consequence. This method, therefore, shows the effect of economic optimization, which is currently the main interest in deriving breeding objectives, on GHG emissions. The second method uses the same model, but now minimizes GHG emissions per kg milk along the chain (i.e. up to the farm gate), to determine the maximum GHG reduction per unit change in milk yield and longevity while maintaining initial labor income and milk production at farm level (i.e. before trait improvement). This method, therefore, determines the full potential of a genetic trait to reduce GHG emissions along the chain, given the boundaries that were set for income and milk production. Results might change when reducing GHG emissions yields additional income. At this moment, however, there is no carbon pricing scheme for agriculture.

2.1 Dairy farm LP model

The dairy farm LP model used is based on Berentsen and Giesen (1995). This static year model includes all relevant activities and constraints that are common to Dutch dairy farms, such as on-farm feed production, purchase of feed products, and animal production including rearing of young stock. The model distinguishes a summer and a winter period regarding feeding. Dietary options include grass from grazing, grass silage, maize silage, and three types of concentrates that differ in protein levels (i.e. standard, medium and high). Nutritional values of the feed ingredients are in Appendix 5.a. Available land can be used as grassland or as maize land. Constraints of the model include fixed resources of the farm (e.g. land area, family labor), links between activities (e.g. fertilizer requirements of grass- and arable land with available nutrients from manure and purchased fertilizers), and environmental policies (e.g. limits to the application of total mineral nitrogen (N) and phosphate (P_2O_5) fertilization). For a more detailed description of the model, see Van Middelaar et al. (2013b).

The central element of the LP model is an average dairy cow from the Holstein Friesian breed, with a given milk production and longevity, calving in February, and conditions representing the dairy cattle of the farm. Feed requirements (energy and protein) and intake capacity of this average cow were determined using the bio-economic model of Groen (1988). The same model was used to determine herd composition and yearly replacement rate, based on the average longevity of the cow. The replacement rate determines the number of young stock that needs to be kept on the farm for yearly replacement of the dairy cows.

The dairy farm model was adapted to future production circumstances to allow exploration of economic and environmental consequences of selective breeding. The generation interval of dairy cattle was assumed to be seven years (CRV, 2012), and, therefore, production circumstances were defined for 2020. In 2015, the milk quota system will be abolished in the EU, and, therefore, no milk quota was assumed. Furthermore, prices of milk components and purchased feed products were adapted based on price prediction for 2020 (KWIN, 2013). Milk price was assumed to be €32.3 per 100 kg milk. Price of purchased maize silage was assumed to be €42 per ton, and for concentrates €180 (standard protein), €208 (medium protein) and €260 (high protein) per ton. Grass yield per hectare was increased by 1.5% per year, based on historical data analysis

(Berentsen et al., 1996). In case of 200 kg N fertilization per ha per year, this implies a grass yield of 72.2 GJ NE_L/ha per year in 2020. The yield of maize silage per hectare was increased by 100 kg DM/year (Rijk et al., 2013), resulting in 108.2 GJ NE_L/ha per year in 2020. For the environmental policies, no changes in limits to the application of N are expected (C.M. Groenestein, Wageningen UR, Wageningen, The Netherlands, personal communication). Therefore, the annual maximum amount of total mineral N/ha is 250 kg for grassland and 140 kg for maize land, and the annual maximum amount for N/ha from animal manure is 250 kg for farms with at least 70% grassland and 170 kg for farms with less than 70% grassland. Limits to the application of P_2O_5 are reduced to an annual maximum of 90 kg P/ha for grassland and 60 kg P/ha for arable land (based on soils with an average phosphate content), according to the new standards for 2020 (Vierde Nederlands Actieprogramma Nitraatrichtlijn, 2009).

2.2 Calculating GHG emissions

We used life cycle assessment (LCA) to calculate emissions of CO_2, CH_4, and N_2O from the different stages along the production chain, up to the moment that milk leaves the farm gate. LCA is an internationally accepted and standardized method to evaluate use of resources and emission of pollutants along the chain (Bauman and Tillman, 2004; Rebitzer et al., 2004). Processes included are the extraction of raw materials to produce farm inputs, the manufacturing and distribution of these inputs, and all processes on the dairy farm. Stages related to transport and processing of milk were assumed to be unaffected by the breeding strategies, and, therefore, not included in the analysis. Methods to calculate annual (i.e. re-current) emissions are described in this section. Methods to calculate non-recurrent emissions from land use change are described in the sensitivity analysis.

Emissions from the production of synthetic fertilizer, pesticides, tap water, and energy sources (gas, diesel, and electricity) were based on Eco-invent (2007), from the production of saw dust on Thomassen et al. (2008), and from the production of concentrates and milk replacer on Vellinga et al. (2013). Emissions from production of concentrates include emissions from the production of inputs (e.g. fertilizers, pesticides, machinery, and energy), direct and indirect N_2O emissions from cultivation, CO_2 emissions from liming and urea fertilization, emissions from drying and processing, and emissions from transport in between stages, up to the farm gate. Emission calculations for the production of purchased maize silage were similar to the calculations that were used for on-farm production of maize silage, because purchased maize silage was assumed to be produced in the Netherlands. Similar to on-farm feed production, yield per ha of purchased feed products (i.e. concentrates and maize silage) were increased by 100 kg DM/year (Rijk et al., 2013), resulting in a decrease in emissions from cultivation per kg ingredient. Emission factors per ton purchased concentrate and maize silage are included in Appendix 5.b.

Emissions of CH_4 from on-farm processes relate to enteric fermentation and to manure management. Enteric CH_4 from dairy cows was calculated based on empirical relations between dry matter intake of feed ingredients and CH_4 emission factors per ingredient. CH_4 emission factors per feed ingredient were based on Vellinga et al. (2013) and are included in Appendix 5.b. For young stock, enteric CH_4 emission was based on IPCC Tier 2 methods and default values, i.e. the average gross energy content of feed is assumed to be 18.45 MJ/kg DM, and 6.5% of the gross energy intake is converted to CH_4 (IPCC, 2006). Emissions of CH_4 from manure management were based on national inventory reports, i.e. 0.746 kg CH_4 per ton manure produced in stables, and 0.110 kg CH_4 per ton manure produced during grazing (De Mol and Hilhorst, 2003).

Emissions of CO_2 from on-farm processes related to the combustion of diesel and gas were based on Eco-invent (2007). Emissions of N_2O from on-farm processes include both direct and indirect N_2O from manure management and from N application to the field, including N from manure, synthetic fertilizers, and crop residues. Indirect N_2O emissions result from N that is removed from the farm via leaching of NO_3^- and volatilization of NH_3 and NO_x (IPCC, 2006). Emissions of N_2O from crop residues were based on IPCC (2006). Other N_2O emissions were based on national inventory reports and are described in more detail by Van Middelaar et al. (2013b).

Different GHGs were summed up based on their equivalence factor in terms of CO_2 equivalents (CO_2e) (100-year time horizon): 1 for CO_2, 25 for CH_4, and 298 for N_2O (Forster et al., 2007). Emissions were calculated per ton FPCM, i.e. milk corrected to a fat percentage of 4.0% and a protein content of 3.3% (Product Board Animal Feed, 2008). After summing up emissions, they were allocated to the different outputs of the farm (i.e. milk and meat from culled calves and cows) based on the relative economic value of these outputs (i.e. economic allocation; KWIN 2008). Economic allocation is used most commonly in LCA studies of livestock products (De Vries and De Boer, 2010).

2.3 Set up of the analysis

The LP model was used to determine the farm plan for a farm in the technical and institutional setting of 2020, and with a cow that has the same characteristics as an average Holstein Friesian cow in 2013. The farm area is 85 ha, which is the estimated size of an average Dutch dairy farm in 2020 (Rabobank, 2009). All manure produced on the farm needs to be applied on the farm, i.e. on grassland and maize land. Traits of the average cow, including milk yield, fat and protein content of the milk, and longevity were based on the CRV database (CRV, 2012), and are included in Table 1. Table 1 also includes information on feed requirements. Based on the assumption that farmers become more efficient in the future, safety margins for true protein digested in the small intestine and for rumen degradable protein balance were set to zero. In previous studies, safety margins for true protein digested in the small intestine were set at 100 g/cow per day, and for rumen degradable protein balance at 200 g/cow per day (Van Middelaar et al., 2013b). The maximum amount of fresh grass intake in summer was set to 12 kg DM/cow per day, based on ad libitum grass intake of cows grazing during day times (Taweel et al., 2004; Abrahamse et al., 2009). Optimization of management variables based on maximizing labor income resulted in the reference scenario.

Table 1. Production traits and feed requirements per cow, and yearly replacement rate of the dairy herd for the reference scenario and after increasing milk yield and longevity with one genetic standard deviation[1].

	Production traits				Feed requirements[4]			Repl. rate
	Milk yield	Fat	Protein	Longevity[3]	Energy	Protein	Intake capacity	
	kg/yr	%	%	# days	GJ NE_L/yr	kg DVE/yr	kg/yr	%
Reference[2]	8758	4.32	3.51	2150	44,553	545	6009	27.0
Incr. milk yield	9445	4.32	3.51	2150	46,961	583	6137	27.0
Incr. longevity	8795[5]	4.31	3.51	2420	44,712	547	6037	22.5

[1] Genetic standard deviation for milk yield is 687 kg/year, and for longevity 270 days (CRV, 2012).

[2] The reference scenario is based on an average Holstein Friesian dairy cow in 2013 (CRV, 2012).

[3] Longevity is defined as the actual age in days.

[4] The diet has to meet a minimum requirement for energy (NE_L, net energy for lactation), a minimum requirement for true protein digested in the small intestine (DVE) and a minimum requirement for rumen degradable protein balance (OEB, not included in the table, for all scenarios set to 0 g/d). In addition, a maximum for intake capacity is included (kg dry matter of the reference feed according to Jarrige (1988).

[5] First lactation yield was the same as in the reference scenario.

To determine the effect of one unit change in milk yield and longevity, each trait was increased with one genetic standard deviation, while keeping the other traits constant. The genetic standard deviation for milk yield of the Holstein Friesian breed in the Netherlands is 687 kg/cow per year (standard deviation applies to milk yield of a mature cow), and for longevity it is 270 days (CRV, 2012). Longevity was defined as the actual age in days. Using the model of Groen (1988), the effect of this change on average production feed requirements, herd composition and replacement rate was determined. Increasing milk yield increased feed requirements (Table 1). Increasing longevity changed herd composition (i.e. more cows in later lactations), and decreased replacement rate and number of young stock. Due to an increase in the number of cows in later lactations, milk yield of the average cow increased and fat content of the milk decreased. Increasing longevity, therefore, indirectly resulted in an increase in feed requirement of dairy cows due to an increased average weight and milk production (Table 1).

The new data on milk yield, feed requirements, and replacement rate for the two scenarios (i.e. increasing milk yield and increasing longevity) were incorporated in the model, and subsequently the impact on GHG emissions was determined by one of the two methods. The first method maximized labor income, and estimated the interrelated consequences for GHG emissions at chain level. The second method minimized GHG emissions per kg milk, and explored the potential of the traits (i.e. milk yield and longevity) to reduce these emissions. For the second method, two additional constraints were required. Labor income and total milk production at farm level were required to be equal or higher than in the reference scenario. Labor income was restricted to estimate the potential of the traits to reduce GHG emissions without sacrificing income. Milk production was restricted to avoid that milk has to be produced somewhere else, which would indirectly mean an increase in GHG emissions from deforestation (i.e. more land required to produce the same amount of milk).

2.4 Deriving economic and GHG values

The economic value represents the change in labor income expressed per cow per year as a result of one genetic standard deviation improvement on milk yield and longevity while keeping the other traits constant. Labor income was defined as gross returns minus variable and fixed costs (including depreciation and interest on fixed assets). Gross returns include revenues of selling of milk and meat (animals). Variable and fixed costs include feed costs, fertilizer costs, costs of building, and other variable and fixed costs. Change in labor income was calculated as 'labor income after change in genetic merit' minus 'labor income before change in genetic merit'. To calculate the economic value per trait, this change was divided by the number of dairy cows present before the change in genetic merit (Groen, 1989).

The GHG value represents the change in GHG emissions expressed per cow per year as a result of one genetic standard deviation improvement in milk yield and longevity while keeping the other traits constant. Whereas deriving economic values is based on maximizing labor income at farm level, deriving GHG values is based on minimizing GHGs per unit of product. Changes in GHG emissions were calculated as 'kg CO_2e per t FPCM before a change in genetic merit' minus 'kg CO_2e per t FPCM after a change in genetic merit'. To calculate the GHG values per trait, this change was multiplied with the FPCM production per cow per year before the change in genetic merit.

2.5 Sensitivity analysis

Two main sources of uncertainty in assessments of GHG emissions of livestock products result from accounting for emissions from land use change (LUC) and handling the link between milk and meat production (i.e. co-product handling) (Zehetmeier et al., 2012; Flysjö et al., 2012). In a sensitivity analysis, we explored the impact of including LUC emissions and using different methods for co-product handling using the results of the previous optimization.

Emissions from LUC

Land use change can have an important impact on GHG emissions and carbon sequestration. In this study, two types of LUC are distinguished: on-farm LUC, i.e. conversion of grassland into maize land and vice versa, and off-farm LUC, i.e. deforestation caused by agricultural expansion.

Conversion of grassland into maize land results in non-recurrent emissions due to a change in soil organic carbon stocks (Vellinga et al., 2011; Van Middelaar et al., 2013b). Conversion of maize land into grassland, on the other hand, can contribute to carbon sequestration. Carbon sequestration was estimated to be 81 t C/ha for permanent grassland, and 41 t C/ha for maize land (see Van Middelaar et al., 2013b). Conversion of grassland into maize land results in an emissions of 163 t CO_2e/ha (see Van Middelaar et al., 2013b for a detailed description of the calculation), whereas conversion of maize land into grassland reduces CO_2 emissions with 147 t/ha (40 t C extra sequestration equals 147 t CO_2).

Deforestation is another type of LUC that results in high amounts of GHGs. Currently, forest is cleared on a large scale to provide land for crop cultivation and pasture. Because of globalization of food and feed markets, it could be stated that every ha of land that is used for commercial production is indirectly responsible for deforestation worldwide (Audsley et al., 2009). To determine the effect of LUC emissions from deforestation on our results, we used the method proposed by Audsley et al. (2009). They divided total GHG emissions from deforestation at world scale for the year 2004 (based on Barker et al., 2007) by the total amount of land used for agricultural production, resulting in one emission factor of 1.43 t CO_2e/ha of land. We applied this emission factor in our study and included on-farm land, and land used to produce purchased feed products, including concentrates and maize silage (Appendix 5.b).

Methods for handling of co-products

We allocated GHG emissions to milk and meat based on economic allocation. Another option to handle co-products is system expansion, implying that we account for changes in GHG emissions resulting from production of additional co-products, i.e. meat from culled calves and cows. In that case, all emissions from processing calves and cows are attributed to milk, whereas emissions related to the production of meat that is replaced by these products, such as pork and chicken, are subtracted. By increasing milk yield and longevity, the ratio of milk over meat production changes, and methodological choices on how to handle co-products might affect results.

In case of system expansion, emissions from processing of calves and cows and from the production of alternative products, such as pork and chicken, need to be calculated. For all meat products, we calculated emissions per kg edible product and included processes up to the gate of the slaughterhouse. For culled dairy cows and young stock older than 12 months, we assumed no further processing and included emissions related to transport and slaughtering only. Surplus calves were assumed to be sold to the white veal industry, which is most common in the Netherlands. Emissions related to processing of these calves were based on H. Mollenhorst (Wageningen UR, Wageningen, The Netherlands, personal communication), and include emissions from feeding, housing, transport and slaughter. Assumption on live weight, amount of edible products per animal, and emissions per kg of edible product from culled calves and cows are included in Appendix 5.c.

Meat from culled calves and cows was assumed to substitute either pork, chicken, or beef from suckler cows, in a ratio of 1:1 based on kg edible product. Emissions from the production of pork, chicken, and beef were based on De Vries and De Boer (2010). De Vries and De Boer (2010) only included emissions up to the farm gate. Emissions related to transport and slaughtering, therefore, were based on the same method that was used for dairy cows. Total GHG emissions along the chain assumed was 7.3 kg CO_2e/kg edible pork, 5.6 kg CO_2e/kg edible chicken, and 23.4 kg CO_2e/kg edible beef. Calculations were made for all three alternatives. In case of pork, for example, we multiplied the amount of edible products from culled calves and cows (i.e. based on the number of animal removed from the farm) with the emission factors presented in Appendix 5.c. Subsequently, we subtracted the same amount of edible products multiplied by the emission factor per kg edible pork. The final result was added to the emissions related to dairy farming, and divided by the total amount of FPCM to determine GHG emission per t FPCM.

3 Results and discussion

3.1 Maximizing labor income

Diets and farm plan

Table 2 shows diets, farm plan, and farm outputs of the reference scenario and the scenarios in which milk yield and longevity were increased with one genetic standard deviation. Results are based on maximizing labor income. For the reference scenario the following results apply. In summer the maximum amount of fresh grass is fed, because grazing is the cheapest way of feeding. Subsequently, maize silage in combination with a small amount of medium protein concentrates is added to meet requirements for energy and rumen degradable protein balance. In winter, the diet contains 2.7 kg DM grass silage per cow per day, based on the amount of grass remained after grazing. Again, maize silage in combination with medium protein concentrates is added to meet requirements for energy and rumen degradable protein balance. The reference scenario has 168 dairy cows, 59.5 ha of grassland and 25.5 ha of maize land. The number of cows is based on the amount of manure that can be applied on the farm according to environmental legislation. In the reference scenario, application standards on the amount of P_2O_5 were restricting (Table 2). The area of grassland is exactly 70%, which is the minimum requirement for farms to comply with the derogation regulation that allows the application of 250 kg N/ha per year from animal manure, instead of 170 kg N/ha per year. Total milk production is 1543 t FPCM/year, i.e. 18.2 t FPCM/ha. Current Dutch dairy farms on sandy soils produce on average about 13.4 t FPCM/ha (FADN, 2010). This means that our reference scenario is more intensive compared to current practice. This results from higher grass and maize yields per ha (i.e. based on predicted increase in yield for 2020), and from application of precision feeding, i.e. ignoring

Table 2. Diets, farm plan, and farm outputs for the reference scenario and after increasing milk yield and longevity with one genetic standard deviation, based on maximizing labor income.

		Reference	Milk yield	Longevity
Diet dairy cows - summer period (kg DM/cow/day)				
Grass (grazed)		12.0	12.0	12.0
Maize silage		8.4	8.9	8.4
Concentrates - medium protein		0.7	1.3	0.7
Diet is restricted by[1]		E,R	E,R	E,R
Diet dairy cows - winter period (kg DM/cow/day)				
Grass silage		2.7	5.0	4.2
Maize silage		8.0	8.9	8.4
Concentrates - medium protein		6.5	4.6	5.0
Diet is restricted by[1]		E,R	E,R	E,R
Farm plan				
Dairy cows	n	168	171	182
Young stock	unit[2]	51	52	46
Grassland 225 kg N/ha	ha	59.5	67.9	67.4
Maize land	ha	25.5	17.1	17.6
Synthetic fertilizer	kg N/ha	107	113	112
	kg P_2O_5/ha	-	-	-
Purchased maize silage	t DM	207	396	381
Purchased concentrates	t DM	247	207	213
Manure application is restricted by[3]		P	aN, P	aN, P
Farm outputs				
Milk	t FPCM	1543	1691	1677
Dairy cows	n	45.3	46.0	40.9
Young stock > 12 months	n	3.9	4.0	3.6
Young stock < 12 months	n	2.1	2.1	1.9
Calves	n	116.8	118.6	135.7
Labor income	€	115,050	135,477	128,765

[1] The diet can be restricted by: E = energy requirements; R = rumen degradable protein balance; T = true protein digested in the small intestine; I = intake capacity.
[2] One unit includes 1 animal < 12 months and 0.96 animal > 12 months.
[3] The intensity of the farm is restricted by the possibility to apply manure. Manure application can be restricted by: tN = total mineral N; aN = N from animal manure; P = P_2O_5.

safety margins for rumen degradable protein balance and true protein digested in the small intestine commonly applied in practice. Labor income in the reference scenario is €115,050/year.

Increasing milk yield by one genetic standard deviation changed the diets and farm plan of the reference scenario (Table 2). The number of cows increased and part of the maize land was changed into grass land. Diets were changed to meet the increasing requirements for energy and protein, and because the area of grassland increased. This resulted from an increase in the number of cows and P_2O_5 application standards being restricting (more P_2O_5 from animal manure can be applied on grassland than on maize land). In the reference scenario, the costs of an increase in grassland at the expense of maize land were higher than the revenues of keeping more cows. After increasing milk yield, the revenues per cow increased, and outweighed the costs of an increase in the area of grassland at the expense of maize land. After increasing milk yield, the number of cows and grassland increased until application standards for N from animal manure became restricting (Table 2). Total milk production at farm level increased to 1691 t FPCM/year, and labor income to €135,477. This is an increase of €122/cow per year.

Increasing longevity by one genetic standard deviation reduced the replacement rate of the dairy herd (from 27.0% in the reference scenario to 22.5% after increasing longevity). Similar to milk yield, this resulted in a situation where maize land was changed into grassland to increase to amount of P_2O_5 that can be applied on the field, and hence the number of dairy cows. Because of the reduced replacement rate, less young stock was kept (Table 2), reducing manure production of the herd. As a result, the number of dairy cows increased to 182. Again, the application standard for N from animal manure limited a further increase of dairy cows. Total milk production at farm level increased to 1677 t FPCM/year, and labor income to €128,765. This is an increase of €82/cow per year.

Greenhouse gas emissions

Table 3 shows GHG emissions for the reference scenario and changes in emissions after increasing milk yield and longevity with one genetic standard deviation. Results are based on maximizing labor income.

In the reference scenario, total GHG emissions per t FPCM added up to 882 kg CO_2e without allocation, and 796 kg CO_2e based on economic allocation. The most important contributor was CH_4 from enteric fermentation (50%). Other important contributors were emissions from manure and from production of concentrates (both 13%). Studies on current milk production systems find values around 1000 kg CO_2e/t FPCM based on economic allocation (Flysjö et al., 2011a; Zehetmeier et al., 2012; Van Middelaar et al., 2013b). Differences are explained by the higher productivity and efficiency representing the technical and institutional setting of 2020 in combination with precision feeding, compared to other studies that represent current production circumstances. These results imply that future dairy farms can reduce their environmental impact in terms of GHG emission per t FPCM when aiming for an increase in efficiency.

Table 3. Greenhouse gas emissions for the reference scenario and the effect of increasing milk yield and longevity with one genetic standard deviation [in kg CO_2e/t FPCM], based on maximizing labor income.

	Reference	Milk yield	Longevity
Animal emissions			
Enteric CH_4 emission dairy cows	372	-5	+3
Enteric CH_4 emission young stock	73	-5	-12
Manure	118	-5	-6
On-farm feed production			
Grassland	67	+6	+6
Maize land	37	-14	-13
Production of farm inputs			
Maize silage	24	+18	+17
Concentrates dairy cows	110	-27	-23
Concentrates young stock	8	-1	-1
Synthetic fertilizer	51	-2	-2
Other inputs	23	-1	0
Total emissions	882	-36	-32
Economic allocation	796	-27	-23

Increasing milk yield by one genetic standard deviation changed GHG emissions from various aspects (Table 3). Changes are expressed in kg CO_2e/t FPCM without allocation. Increasing milk yield per cow reduced emissions per t FPCM by diluting emissions related to maintenance and young stock. In addition, emissions changed because of changes in diets and farm plan. Emissions from the production of purchased maize silage increased (18 kg CO_2e/t FCPM), because the amount of maize silage in the diets increased while the production of on-farm maize silage decreased. Given the decline in amount of concentrate required per t FPCM, emissions from concentrate production decreased (27 kg CO_2e/t FPCM). Increasing milk yield by one genetic standard deviation decreased total GHG emissions per t FPCM by 36 kg CO_2e using no allocation, and by 27 kg CO_2e using economic allocation. Based on economic allocation, the GHG value of milk yield is 247 kg CO_2e/cow per year.

Increasing longevity by one genetic standard deviation mainly affected emissions related to young stock and production of feed for dairy cows. Due to a lower replacement rate, emissions related to young stock (enteric CH_4 and emissions from concentrate production) decreased by 13 kg CO_2e/t FPCM. Due to a change in the diets of dairy cows towards more roughage and less concentrates, emissions from production of grass and maize silage increased by 10 kg CO_2e/t FPCM (including on- and off farm production), whereas emissions from production of concentrates decreased by

23 kg CO_2e/t FPCM (Table 3). Increasing longevity by one genetic standard deviation decreased total GHG emissions per t FPCM by 32 kg CO_2e using no allocation, and by 23 kg CO_2e using economic allocation. Based on economic allocation, the GHG value of longevity is 210 kg CO_2e/cow per year, which is 15% lower than the GHG value of milk yield.

3.2 Minimizing GHG emissions

Diets and farm plan

Table 4 shows diets, farm plan, and farm outputs of the reference scenario and the scenarios in which milk yield and longevity were increased by one genetic standard deviation. Results for the scenarios with increased milk yield and longevity are based on minimizing GHG emissions per kg milk. In all scenarios, labor income and total milk production had to level or exceed the amounts of the reference scenario. The reference scenario and its results, therefore, are exactly the same as for the method based on maximizing labor income (Table 2).

Increasing milk yield by one genetic standard deviation and, subsequently, minimizing GHG emissions per t FPCM affected the diet and farm plan. Labor income and total milk production were not restricting, which means that the optimal solution within the feasible region was determined by other constraints. The level of N fertilization on grassland reduced from 225 kg N/ha per year to 200 kg N/ha per year. A reduction in N fertilization on grassland reduced emissions from cultivation per kg DM grass and grass silage, whereas enteric CH_4 emissions increased (Appendix 5.b). Within the limits of environmental legislations on the application of manure, a fertilization level of 200 kg N/ha per year resulted in the lowest GHG emissions per t FPCM. The amount of concentrates in the diets was minimized. In summer, a decrease in kg DM grass per cow per day allowed for an increase in kg DM maize silage per cow per day, and a maximum roughage uptake within the limiting intake capacity. Also in winter the maximum amount of roughage was fed. In both diets, high protein concentrates were added to fulfill requirements for rumen degradable protein balance. Although concentrates result in a lower enteric CH_4 emission than grass and grass silage, emissions during production are much higher (Appendix 5.b). The number of cows was restricted by P application standards and the area of grassland (because application standards are higher for grassland than for maize land). The area of grassland was determined by the amount of grass and grass silage in the diet, and hence, by the maximum intake capacity of the cow. With total milk production as a minimum constraint, the number of cows increased as long as the increase in grass and grass silage was fully consumed,

Table 4. Diets, farm plan, and farm outputs for the reference scenario and after increasing milk yield and longevity with one genetic standard deviation, based on minimizing GHG emissions.

		Reference	Milk yield	Longevity
Diet dairy cows - summer period (kg DM/cow/day)				
Grass (grazed, 200 kg N/ha)		-	11.7	12.0
Grass (grazed, 225 kg N/ha)		12.0	-	-
Maize silage		8.4	9.3	8.4
Concentrates - medium protein		0.7	-	-
- high protein		-	1.3	0.8
Diet is restricted by [1]		E,R	E,R,I	E,R
Diet dairy cows - winter period (kg DM/cow/day)				
Grass silage (200 kg N/ha)		-	4.5	5.3
Grass silage (225 kg N/ha)		2.7	-	-
Maize silage		8.0	11.2	10.4
Concentrates - medium protein		6.5	-	-
- high protein		-	2.9	2.3
Diet is restricted by[1]		E,R	E,R,I	E,R,I
Farm plan				
Dairy cows	n	168	157	178
Young stock	unit[2]	51	48	45
Grassland 200 kg N/ha	ha	-	62.4	72.5
Grassland 225 kg N/ha	ha	59.5	-	-
Maize land	ha	25.5	22.6	12.5
Synthetic fertilizer	kg N/ha	107	92	95
	kg P_2O_5/ha	-	-	-
Purchased maize silage	t DM	207	338	512
Purchased concentrates	t DM	247	141	117
Manure application is restricted by[3]		P	P	P
Farm outputs				
Milk	t FPCM	1543	1558	1640
Dairy cows	n	45.3	42.4	40.0
Young stock > 12 months	n	3.9	3.7	3.5
Young stock < 12 months	n	2.1	1.9	1.8
Calves	n	116.8	109.3	132.7
Labor income	€	115,050	127,301	120,428

[1] The diet can be restricted by: E = energy requirements; R = rumen degradable protein balance; T = true protein digested in the small intestine; I = intake capacity.

[2] One unit includes 1 animal < 12 months and 0.96 animal > 12 months.

[3] The intensity of the farm is restricted by the possibility to apply manure. Manure application can be restricted by: tN = total mineral N; aN = N from animal manure; P = P_2O_5.

and P application standards were not restricting. After increasing milk yield, this balance was reached at 157 cows, which is lower than the number of cows in the reference scenario. Total milk production increased from 1543 t FPCM to 1558 t FPCM/year, and labor income from €115,050 to €127,301/year. This is an increase of €73/cow per year.

Increasing longevity resulted in a similar strategy as increasing milk yield. N fertilization on grassland reduced to 200 kg N/ha per year, and the amount of concentrates was minimized. The number of cows increased to 178; a further increase was restricted by P application standards in combination with limits to the amount of grass and grass silage in the diets. Total milk production increased to 1640 t FPCM/year, and labor income to €120,428/year. This is an increase of €32/cow per year.

Greenhouse gas emissions

Table 5 shows GHG emissions of the reference scenario and changes in emissions after increasing milk yield and longevity with one genetic standard deviation. Results are based on minimizing GHG emissions per kg milk. For the reference scenario, results are exactly the same as for the method based on maximizing labor income (Table 3).

Table 5. Greenhouse gas emissions for the reference scenario and the effect of increasing milk yield and longevity with one genetic standard deviation [in kg CO_2e/t FPCM], based on minimizing GHG emissions.

	Reference	Milk yield	Longevity
Animal emissions			
Enteric CH_4 emission dairy cows	372	-2	+11
Enteric CH_4 emission young stock	73	-5	-12
Manure	118	-5	-5
On-farm feed production			
Grassland	67	-1	+8
Maize land	37	-5	-20
Production of farm inputs			
Maize silage	24	+15	+32
Concentrates dairy cows	110	-48	-62
Concentrates young stock	8	-1	-1
Synthetic fertilizer	51	-8	-8
Other inputs	23	0	0
Total emissions	882	-60	-59
Economic allocation	796	-49	-48

After increasing milk yield per cow, N fertilization on grassland was decreased and the diets of the dairy cows were changed. Per t FPCM, emissions related to on-farm feed production, enteric fermentation, and production of various farm inputs decreased. Most important was the reduction in emissions from concentrates production (48 kg CO_2e/t FPCM without allocation). Emissions from the production of purchased maize silage were the only emissions that increased (15 kg CO_2e/t FPCM). Overall, emissions per t FPCM decreased by 60 kg CO_2e using no allocation, and by 49 kg CO_2e using economic allocation. Based on economic allocation, the GHG value of milk yield is 453 kg CO_2e/cow per year.

After increasing longevity, the number of young stock decreased, N fertilization on grassland decreased and the diets of dairy cows changed. Per t FPCM, emissions related to young stock, on-farm feed production, and production of synthetic fertilizer decreased. Most significant was again the reduction in emissions from concentrate production. Emissions from production of purchased maize silage and enteric fermentation of dairy cows increased. Overall, emissions per t FPCM decreased by 59 kg CO_2e using no allocation, and by 48 kg CO_2e using economic allocation. Based on economic allocation, the GHG value of longevity is 441 kg CO_2e/cow per year, which is 3% lower than the GHG value of milk yield.

3.3 Sensitivity analysis

Emissions from land use change

It is important to realize that changing genetic traits does not only affect annual emissions, but can also affect non-recurrent emissions related to on-farm LUC. After increasing milk yield and longevity, maize land was changed into grassland contributing to CO_2 sequestration. Using method 1, this change in land use resulted in sequestration of 667 kg CO_2e/t FPCM per unit change of milk yield, and 636 kg CO_2e/t FPCM per unit change of longevity. Using method 2, sequestration was 252 kg CO_2e/t FPCM for milk yield and 1062 kg CO_2e/t FPCM for longevity. Results show that both traits result in CO_2 sequestration due to on-farm LUC. Using method 1, the importance of milk yield relative to longevity increases, whereas for method 2 the importance of longevity increases. As opposite to the annual emissions that were used to calculate GHG values, emissions from on-farm LUC are non-recurrent and, therefore, cannot be included in the GHG value of traits.

Emissions from deforestation were calculated for the year 2004 (Barker et al., 2007; Audsley et al., 2009). Under the assumption that the rate of deforestation in the year 2004 can be used as an indicator for other years, emissions from deforestation were treated as re-current emissions and included in the GHG values. Using method 1, including emissions from deforestation increased the GHG value of milk yield from 247 to 260 kg CO_2e/cow per year and of longevity from 210 to 219 kg CO_2e/cow per year. This increase in GHG values resulted from a decrease in land use per t FPCM. Using method 2, the GHG value of milk yield decreased from 453 to 435 kg CO_2e/cow per year and of longevity from 441 to 406 kg CO_2e/cow per year. Including emissions from deforestation did not affect the relative importance of the traits and the absolute change in GHG values was limited. When the pressure on land use and the importance of deforestation increases, it can become relevant to include emissions from deforestation in GHG values, and in the optimization procedure. Because of high uncertainty and variation in calculating emissions from deforestation, and because they differ in nature from annual emissions at farm-level, GHG values should be presented with and without emissions from deforestation (Van Middelaar et al., 2013a; Flysjö et al., 2012).

Methods for handling of co-products

Table 6 shows the GHG values of milk yield and longevity using different methods for handling of co-products. Results show that methodological choices affect GHG values.

GHG values of milk yield and longevity change when using system expansion instead of economic allocation, and vary for different alternative products (Table 6). The absolute change in GHG values was the same for method 1 and 2. With system expansion, the difference in GHG values between milk yield and longevity are larger than with economic allocation. When longevity increases, fewer cows but more calves are culled. Because calves have a higher emission factor per

Table 6. GHG values of milk yield and longevity [kg CO_2e/cow per year] using different methods for handling co-products, based on maximizing labor income and minimizing GHG emissions.

		Maximizing labor income		Minimizing GHG emissions	
		Milk yield	Longevity	Milk yield	Longevity
No allocation or system expansion		326	291	552	544
Economic allocation		247	210	453	441
System expansion	- pork	303	186	529	439
	- chicken	321	201	547	453
	- beef	133	50	359	302

kg of edible product than cows, increasing longevity increased emissions related to processing of culled animals, and hence resulted in a lower GHG value than an increase in milk yield.

Of the three alternative products, beef resulted in the lowest GHG values. Producing beef results in high emissions. Under the assumption that meat from culled calves and cows replaces beef from suckler cows, these emissions are subtracted from the CFP of milk. An increase in milk yield or longevity means that per kg FPCM, less meat is produced that can substitute beef, and hence, fewer emissions are subtracted per kg FPCM. When fewer emissions are subtracted, total reduction per kg FPCM is lower, and, therefore, the GHG value. Overall, these results indicate that increasing milk yield results in a higher GHG value than longevity when co-products are included in the analyses.

4 General discussion

Using method 1, increasing milk yield with 687 kg/cow per year reduced GHG emissions by 4.1% to 6.8% (no allocation or system expansion). Bell et al. (2011) found a reduction of 9.5% to 13.2% per genetic standard deviation improvement of 1241 kg ECM/cow per year. Bell et al. (2011), however, increased milk yield while maintaining the same feed intake, which makes direct comparison difficult. For longevity, we found a reduction in GHG emissions of 3.6% to 6.7% when increasing the average number of lactations from 3.25 to 3.90 (no allocation or system expansion). Wall et al. (2010) found a reduction of 4.3% when increasing longevity from 3.0 to 3.5 lactations, but only included CH_4 and N_2O emissions at farm level. Both Wall et al. (2010) and Bell et al. (2011) did not optimize farm management with changing levels of genetic traits, which can lead to over- or underestimation. They also used different methods to calculate emissions. To calculate enteric CH_4 emissions, for example, we used feed specific emission factors derived from mechanistic modeling techniques. This method is found to be more precise than IPCC Tier 1 and 2 methods that were used by Wall et al. (2010) and Bell et al. (2011) (Kebreab et al., 2008; Alemu et al., 2011). Harmonization of methods to calculate GHG values is important when breeding organizations want to include environmental performance into their breeding goal.

Based on the results presented in this study, milk yield is more important than longevity when both economics and reducing GHG emission are relevant (method 1: the economic value of milk yield is 49% higher than the value of longevity and the GHG value is 18% higher). If only reducing GHG emissions is relevant, milk yield and longevity are approximately equally important (method 2: the GHG value of milk yield is only 3% higher than the value of longevity). In that case, however, milk yield still has a higher economic value than longevity (€73/cow per year compared to €32/cow per year). This indicates that in a situation where labor income becomes restricting, milk yield could result in a higher GHG value than longevity, because more money is available to implement improvement options.

Results were based on a situation without output limitations, which allows for a change in the number of cows. In addition, management factors such as on-farm roughage production and purchases of feeds changed with a change in trait. As a result, GHG values not only represent the direct impact of a change in traits, but also the indirect impact due to changes in number of cows and management factors. Because these factors are variable, and can be influenced by the traits of the cow, including both direct and indirect impacts of a change in traits was assumed to be most accurate for estimating GHG values.

Differences in diets and farm plan between method 1 and 2 show that dairy farmers are likely to change their management when standards to reduce GHG emissions are introduced. Compared to method 1, method 2 resulted in a reduction in N fertilization on grassland, and a reduction in the amount of concentrates. Results of both methods are influenced by the reference scenario, including farm size and environmental policies. In addition, production characteristics of the average cow in the reference scenario might have influenced results. This means that changing the constraints of the model, or changing aspects of the reference scenario might change results (Berentsen and Tiessink, 2003). Before using GHG values to define the breeding goal, the effect of such changes should be considered. Furthermore, a wider sensitivity analysis regarding prices and emission factors is required to verify the relative value of traits.

Methods used to calculate relative GHG values of genetic traits included not only emissions at farm level, but also emissions from the production of farm inputs (e.g. purchased feed). Although the dairy sector is not responsible for reducing emissions from industrial processes related to the production of farm-inputs, it is important to consider these off-farm emissions in the analysis. Excluding these emissions might yield strategies that reduce emissions at farm level, but increase emissions during production of farm-inputs (i.e. pollution swapping). By purchasing feed ingredients with a low environmental impact, dairy farmers can contribute to reduced emissions. In addition, this approach is consistent with accounting for costs of farm inputs in the calculation of economic values.

To illustrate the two methods presented in this study, we only considered the effect of an increase in milk yield and longevity. Correlations between traits were not considered because we assumed that these traits will be included in the breeding goal. When calculating GHG values of longevity, we did not account for the potential loss in genetic gain through replacement heifers being genetically superior to older cows. Including this potential loss would decrease GHG values of longevity.

The need to include GHG values into breeding goals is currently limited, but might increase because of the increasing concerns about GHG emissions. Relative GHG values can be used in a similar way as economic values, i.e. to determine which traits are most important when aiming for a certain goal (in this case minimizing GHG emissions).

Results show that N_2O emissions related to N fertilization of grassland are negatively correlated with CH_4 emissions related to enteric fermentation of grass and grass silage from this grassland. Thus, reducing N fertilization on grassland reduces N_2O emissions, but increases enteric CH_4 emissions. Hence, the net effect on GHG emissions per t milk is minimal and reducing N fertilization on grassland below 200 kg N/ha in a situation similar to our study does not pay off.

5 Conclusions

Both methods presented in this study provide insight into the GHG value of genetic traits, but give different information. Which method to use depends on the objective of the dairy industry and the standards they have to meet. Method 1 shows the GHG value of traits in situations where maximizing labor income is the main objective, which relates to current practice. Method 2 shows the full potential of traits to reduce GHG emissions given the boundaries that were set for income and milk production, and relates to a situation where lowering emissions becomes more important than increasing income. Calculating GHG values of milk yield and longevity shows that the full potential of both traits to reduce GHGs is about twice as high as the reduction based on maximizing labor income. In addition, milk yield has a higher GHG value than longevity, especially when focus is on maximizing labor income. Including emissions from LUC, and using different methods for handling co-products generally changed results further in favor of milk yield. Results can be used by breeding organizations that want to include GHG values in their breeding goal. To verify GHG values, the effect of prices and emissions factors should be considered, as well as the potential effect of variation between farm types.

Chapter 6

Impact of farm characteristics on relative values of genetic traits in dairy cows to reduce greenhouse gas emissions along the chain

C.E. van Middelaar[a], P.B.M. Berentsen[b], J. Dijkstra[c], J.A.M. van Arendonk[d], and I.J.M. de Boer[a]

[a] Animal Production Systems group, Wageningen University, Wageningen, the Netherlands

[b] Business Economics group, Wageningen University, Wageningen, the Netherlands

[c] Animal Nutrition group, Wageningen University, Wageningen, the Netherlands

[d] Animal Breeding and Genomics Centre, Wageningen University, Wageningen, the Netherlands

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Abstract

Breeding has potential to reduce greenhouse gas (GHG) emissions from dairy farming. Evaluating the impact of one unit change (i.e. one genetic standard deviation improvement) in genetic traits on GHG emissions along the chain provides insight into the relative importance of genetic traits to reduce GHG emissions. Relative GHG values of genetic traits, however, might depend on feed-related farm characteristics. The objective of this study was to evaluate the impact of feed-related farm characteristics on GHG values, by comparing the values of milk yield and longevity for an efficient and less efficient farm. GHG values of milk yield and longevity were calculated by using a whole-farm model and two different optimization methods. The first method optimizes farm management before and after a change in genetic trait by maximizing labor income; the impact on GHG emissions (i.e. from production of farm inputs up to the farm gate) was considered as a side effect. The second method optimizes farm management after a change in genetic trait by minimizing GHG emissions per kg milk, while maintaining labor income and milk production at least at the level before the change in trait; the impact on labor income was considered as a side effect. Results revealed that the impact of feed-related farm characteristics was large when GHG values were calculated based on maximizing labor income. On the less efficient farm, GHG values of milk yield and longevity were respectively 279 and 143 kg CO_2 equivalents (CO_2e)/unit change per cow per year, whereas on the efficient farm these values were 247 and 210 kg CO_2e/unit change per cow per year. Hence, the GHG value of milk yield relative to the GHG value of longevity decreased with increase in farm-efficiency. Based on minimizing GHG emissions, GHG values of milk yield and longevity were respectively 538 and 563 kg CO_2e/unit change per cow per year on the less efficient farm, and 453 and 441 kg CO_2e/unit change per cow per year on the efficient farm. Hence, on each level of efficiency, the relative importance of both traits was equal but the absolute impact of a change was smaller on the efficient farm. The impact of feed-related farm characteristics on the relative importance of traits to reduce GHG emissions can be great, particularly when optimizing farm management based on maximizing labor income.

1 Introduction

Dairy cattle breeding has potential to reduce greenhouse gas (**GHG**) emissions from dairy farming (e.g. Hayes et al., 2013). Breeding for increased animal productivity, for example, reduces the number of animals needed to produce the same amount of product, and is seen as an important strategy to reduce GHG emissions (Hristov et al., 2013b). In contrast with most other type of management strategies, such as dietary changes, breeding is a long-term strategy, with permanent and cumulative effects. This implies that a good planning is essential when deciding on a breeding strategy.

Most studies that explored breeding strategies to reduce GHG emissions focus on reducing the emission of enteric methane (CH_4) (Bell et al., 2010; De Haas et al., 2011; Hansen Axelsson et al., 2013). Genetic improvement, however, can affect the whole farm, including the diet of dairy cows and on-farm feed production (Wall et al., 2010; Bell et al., 2010). As a result, not only enteric CH_4, but also other GHG emissions related to characteristics of cows and activities on the dairy farm might change. In addition, a strategy can affect the type and amount of purchased products, such as feed and fertilizers. Hence, GHG emissions related to upstream processes might change as well. Evaluating the impact of a genetic improvement, therefore, requires an integrated approach that accounts for changes in farm management and includes all GHG emissions along the chain, i.e. from production of farm inputs up to the farm gate (Wall et al., 2010; Van Middelaar et al. 2013b; 2014).

Evaluating the impact of one unit change in genetic traits on GHG emissions along the chain (i.e. from production of farm inputs up to the farm gate) provides insight into the potential impact of individual traits to reduce GHG emissions (Van Middelaar et al., 2014). Such 'GHG values' can be used to implement environmental performance of traits in breeding programs (Wall et al., 2010). Van Middelaar et al. (2014) examined two methods to calculate GHG values of genetic traits by using a whole farm optimization model in combination with a life cycle approach (i.e. including all GHG emissions up to the farm gate). The first method optimized farm management before and after a change in genetic trait by maximizing labor income; the impact on GHG emissions was considered as a side effect. The second method optimized farm management after a change in genetic merit by minimizing GHG emissions per kg milk, while maintaining labor income and milk production at least at the level before the change in trait. The impact of methods were illustrated for one genetic standard deviation improvement in milk yield and in longevity. It was shown that GHG values of both traits were about twice as high when focus was on minimizing

GHG emissions than when focus was on maximizing labor income. In addition, GHG values of milk yield were larger than GHG values of longevity, especially when focus was on maximizing labor income.

The GHG values calculated by Van Middelaar et al. (2014) applied to one typical dairy farm in 2020, with a high efficiency concerning feed utilization and feed production at farm level. High efficiency in feed utilization was obtained by ignoring safety margins for true protein digested in the small intestine (DVE) and for rumen degradable protein balance (RDPB). Such an increase in efficiency might be reached by precision feeding. High efficiency in on-farm feed production was obtained by increasing grass and maize yields per hectare based on historical data analysis to estimate yields for 2020. Several studies have shown, however, that the environmental impact of milk production varies between farms, and that this variation is often feed-related (Thomassen et al., 2009; Meul et al., 2014). Examples of feed-related farm characteristics causing variation in GHG emissions are type and amount of feed used per cow, level of crop yield per ha, and level of nitrogen application for on-farm roughage production (Thomassen et al. 2009; Meul et al., 2014). It is unclear how GHG values of genetic traits depend on feed-related farm characteristics (i.e. no precision feeding, lower yield per ha).

The objective of this paper is to explore the robustness of GHG values to assumptions on feed-related farm characteristics. The GHG values of milk yield and longevity were calculated for a less efficient farm and compared to those calculated for an efficienct farm by Van Middelaar et al. (2014). The less efficient farm does not apply precision feeding and has a lower grass and maize yield per ha than the efficient farm.

2 Methods

Methods used to calculate GHG values of milk yield and longevity are described in detail in Van Middelaar et al. (2014). The following paragraphs include a short description of the most important aspects of the model and a description of the analysis to determine GHG values. Under set up of the analysis, differences between the efficient and less efficient farm are explained.

The aggregate genotype for our analysis consisted of milk yield and longevity. Genetic variation in other traits was ignored. The relative GHG value of a genetic trait represents the impact of one unit change on GHG emissions at chain level while keeping the other trait constant. The chain level included all processes related to milk production, from the production of raw materials to produce farm inputs (e.g. feed and fertilizers) up to the moment the milk leaves the farm gate. Results (income and GHG emissions) for the optimized farm before and after one standard deviation improvement of milk yield (longevity) were determined using a dairy farm linear programming (LP) model. Two methods were used for optimization. Method 1 optimized farm management by maximizing labor income, while the impact on GHG emissions is considered as a side effect. This method is similar to the method that is generally used to calculate economic values. Economic values form the basis for current breeding goals for dairy breeding. Method 2 optimized farm management by minimizing GHG emissions per kg milk, while maintaining not only labor income but also milk production from the herd at least at the level before the change trait. This method, therefore, shows what can be achieved if breeding results are fully directed at reducing GHG emissions within the constraints set for income and milk production.

2.1 Dairy farm LP model

The dairy farm LP model is based on Berentsen and Giesen (1995), and adapted to 2020 to allow exploration of economic and environmental consequences of dairy cattle breeding. The model includes all relevant activities and constraints that are common to Dutch dairy farms (e.g. on-farm feed production, purchase of feed products, animal production, environmental policies). It distinguishes a summer and a winter period regarding feeding, and includes different dietary options (i.e. grass from grazing, grass silage, maize silage, and three types of concentrates that differ in protein levels). The maximum amount of fresh grass intake in summer was set to 12 kg DM/cow per day, based on ad libitum grass intake of cows grazing during day times (Abrahamse et al., 2009). The central element of the model is an average dairy cow from the Holstein Friesian breed, with a given milk production and calving in February. Feed requirements (energy and protein) and intake capacity of this average cow were determined using the bio-economic model of Groen (1988). This model was used also to determine herd composition and yearly replacement rate, based on the average longevity of the cow. Adaptations of the LP model to future production circumstances include abolition of the milk quota, changes in prices of milk components and purchased feed products, increasing yield per ha, and changes in environmental policies regarding application of fertilizers and manure on the farm. The environmental policies include an annual maximum supply of total mineral N/ha of 250 kg for grassland and 140 kg for maize land, and an annual maximum supply for N/ha from animal manure of 250 kg for farms with at least 70 % grassland and 170 kg for farms with less than 70 % grassland. Regarding phosphate (P_2O_5), the annual maximum supply is 90 kg P_2O_5/ha for grassland and 60 kg P_2O_5/ha for arable land (based on soils with an average phosphate content). Constraints were set such that all manure produced on the farm had to be applied on the farm, i.e. on grassland and maize land.

2.2 Calculating GHG emissions

Life cycle assessment (**LCA**) was used to calculate emissions of CO_2, CH_4, and N_2O from the different stages along the production chain, from production of farm inputs up to the moment that milk leaves the farm gate. Processes included are the extraction of raw materials to produce farm inputs (e.g. energy sources, fertilizers, feed), the manufacturing and distribution of these inputs, and all processes on the dairy farm involved in production of feed and milk. Emissions of CO_2, CH_4, and N_2O were combined based on their equivalence factor in terms of CO_2 equivalents (CO_2e) (100-year time horizon): 1 for CO_2, 25 for CH_4, and 298 for N_2O (Forster et al., 2007). Emissions were calculated per ton fat-and-protein corrected milk (FPCM), i.e. milk corrected to a fat percentage of 4.0 % and a protein content of 3.3 % (Product Board Animal Feed, 2008). After accumulating emissions, they were allocated to the output of milk and meat from culled calves and cows based on the prices (incl. VAT) (KWIN, 2008) of these outputs (i.e. economic allocation). Economic allocation is most commonly used in LCA studies of livestock products (De Vries and De Boer, 2010).

2.3 Set up of the analysis

The LP model was used to determine the farm plan of the farm in the technical and institutional setting of 2020, and with a cow that has the same characteristics as an average Holstein Friesian cow in 2013. The farm area is 85 ha, which is the estimated size of an average Dutch dairy farm in 2020 (Rabobank, 2009). Traits of the average cow (milk yield, fat and protein content of the milk, and longevity) were based on the CRV database (CRV, 2012), and are included in Table 1. Table 1 also includes information on feed requirements of the cow (Groen et al., 1988). Optimization of farm management by maximizing labor income resulted in the reference scenario, i.e. the scenario before genetic improvement.

To determine the effect of one unit change in milk yield and longevity, each trait was increased with one genetic standard deviation, while keeping the other traits constant. The genetic standard deviation for milk yield is 687 kg/cow per year (standard deviation applies to milk yield of a mature cow), and for longevity 270 days (CRV, 2012). Longevity is defined as the actual age in

Table 1. Production traits and feed requirements per cow, and yearly replacement rate of the dairy herd for the reference scenario and after increasing milk yield and longevity with one genetic standard deviation[1].

	Production traits				Feed requirements[4]			Repl. rate
	Milk yield	Fat	Protein	Longevity[3]	Energy	Protein	Intake capacity	
	kg/yr	%	%	# days	GJ NE_L/yr	kg DVE/yr	kg/yr	%
Reference[2]	8758	4.32	3.51	2150	44,553	545	6009	27.0
Incr. milk yield	9445	4.32	3.51	2150	46,961	583	6137	27.0
Incr. longevity	8795[5]	4.31	3.51	2420	44,712	547	6037	22.5

[1] Genetic standard deviation for milk yield is 687 kg/year, and for longevity 270 days (CRV, 2012).
[2] The reference scenario is based on an average Holstein Friesian dairy cow in 2013 (CRV, 2012).
[3] Longevity is defined as the actual age in days.
[4] The diet has to meet a minimum requirement for energy (NE_L, net energy for lactation), a minimum requirement for true protein digested in the small intestine (DVE) and a minimum requirement for rumen degradable protein balance (OEB, not included in the table, for all scenarios set to 0 g/d). In addition, a maximum for intake capacity is included (kg dry matter of the reference feed according to Jarrige (1988).
[5] First lactation yield was the same as in the reference scenario.

days when the cow leaves the farm. Using the model of Groen (1988), the effect of this change on average production, feed requirements, herd composition and replacement rate was determined (Table 1).

The new data on milk yield, feed requirements, and replacement rate for the two scenarios (i.e. increased milk yield and increased longevity) were incorporated in the model, and subsequently the impact on GHG emissions was determined by one of the two objectives, i.e. maximizing labor income (method 1), or minimizing GHG emissions per kg milk (method 2).

2.4 Differences between the efficient and less efficient farm

Van Middelaar et al. (2014) made two important assumptions regarding feed-related farm characteristics. First, grass and maize yields for 2020 were based on current yields plus an average annual yield increase according to historical data analysis (Berentsen et al., 1996; Rijk et al., 2013). Grass yield in 2020 was assumed to be 72.2 GJ NE_L/ha per year (based on a fertilization level of 200 kg N/ha), and maize yield 108.2 GJ NE_L/ha per year. Second, safety margins for DVE and for RDPB were eliminated.

To explore the sensitivity of GHG values of genetic traits, two changes were made to lower the efficiency of the farm compared to that of the efficient farm described by Van Middelaar et al.

(2014): grass and maize yield per ha per year were set at 95% of the yields in the efficient farm, and safety margins for DVE and RDPB were set at 100 and 200 g/cow per day, respectively, as compared to zero in the efficient farm. Safety margins used in the less efficient farm correspond with safety margins used in studies focusing on current systems (Van Middelaar et al., 2013b).

2.5 Deriving economic and GHG values

Economic values represent the change in labor income from the farm, expressed per cow per year as a result of one genetic standard deviation increase of a trait while keeping the other traits constant. Change in labor income was calculated as 'labor income after change in genetic trait' minus 'labor income before change in genetic trait'. Subsequently, this change was divided by the number of dairy cows present before the change in genetic trait (Groen, 1989).

The GHG values represent the change in GHG emissions along the chain, expressed per cow per year as a result of one genetic standard deviation increase of a trait while keeping the other traits constant. Changes in GHG emissions were calculated as 'kg CO_2e per t FPCM before a change in genetic trait' minus 'kg CO_2e per t FPCM after a change in genetic trait'. Subsequently, this change was multiplied with the FPCM production per cow per year in tons before the change in genetic trait.

3 Results and discussion

In each paragraph we first discuss results of the less efficient farm, and subsequently compare these to results of the efficient farm. Results of the efficient farm are included in Chapter 5, Table 2, p 93 (method 1) and in Chapter 5, Table 4, p 97 (method 2).

3.1 Maximizing labor income (method 1)

Diets and farm plan

Table 2 shows the diets, farm plan, and farm outputs of the reference scenario and the scenarios in which milk yield or longevity was increased with one genetic standard deviation.

For the reference scenario the following results apply. In summer, the maximum amount of fresh grass is fed, which is the cheapest way of feeding. Maize silage and medium protein concentrates

Table 2. Diets, farm plan, and farm outputs of the less efficient farm, for the reference scenario and after increasing milk yield and longevity with one genetic standard deviation, based on maximizing labor income.

		Reference	Milk yield	Longevity
Diet dairy cows - summer period (kg DM/cow/day)				
Grass (grazed)		12.0	12.0	12.0
Maize silage		5.1	5.6	5.1
Concentrates - medium protein		3.7	4.3	3.7
Diet is restricted by[1]		E,R	E,R	E,R
Diet dairy cows - winter period (kg DM/cow/day)				
Grass silage		4.9	7.2	4.9
Maize silage		5.3	6.3	5.3
Concentrates - medium protein		7.1	5.2	7.2
Diet is restricted by[1]		E,R	E,R,I	E,R
Farm plan				
Dairy cows	n	145	147	150
Young stock	unit [2]	44	45	38
Grassland 225 kg N/ha	ha	60.6	68.0	60.7
Maize land	ha	24.4	17.0	24.3
Synthetic fertilizer	kg N/ha	100	105	100
	kg P_2O_5/ha	-	-	-
Purchased maize silage	t DM	-	155	-
Purchased concentrates	t DM	310	279	320
Manure application is restricted by [3]		P	P	P
Farm outputs				
Milk	t FPCM	1336	1459	1380
Dairy cows	n	39	39.7	33.7
Young stock > 12 months	n	3.4	3.5	2.9
Young stock < 12 months	n	1.8	1.8	1.5
Calves	n	101	102.4	111.7
Labor income	€	89,885	107,781	101,010

[1] The diet can be restricted by: E = energy requirements; R = rumen degradable protein balance; T = true protein digested in the small intestine; I = intake capacity.
[2] One unit includes 1 animal < 12 months and 0.96 animal > 12 months.
[3] The intensity of the farm is restricted by the possibility to apply manure. Manure application can be restricted by: tN = total mineral N; aN = N from animal manure; P = total P_2O_5.

were added to meet requirements for energy and RDPB. In winter, the diet contained 4.9 kg DM grass silage per cow per day, based on the amount of grass remaining after grazing during summer. Maize silage and medium protein concentrates were added to meet requirements for energy and RDPB. The reference scenario had 145 dairy cows, 60.6 ha of grassland and 24.4 ha of maize land. The number of cows reached the level where maize silage had to be purchased in case of a further increase. Division of the land between grass- and maize land was determined by energy production per ha and P_2O_5 application standards. Maize land had a higher energy production per ha than grassland, but lower P_2O_5 application standards. The result is an area of grass and maize land that resulted in the highest number of cows (based on P_2O_5 application standards) without purchasing maize silage. In the reference scenario, total milk production at farm level was 1336 t FPCM/year, i.e. 15.7 t FPCM/ha per year, and labor income €89,885/year.

Increasing milk yield changed the diets and farm plan (Table 2). The number of cows increased from 145 to 147, and amount maize land reduced in favor of grassland. Diets were changed to meet the increased requirements for energy and protein per cow, and because the area of grassland increased. In the reference scenario, reducing maize land in favor of grassland to allow for an increase in the number of cows was not beneficial anymore when maize silage had to be purchased. After increasing milk yield, the revenues per cow increased, and reducing maize land in favor of grassland to increase the number of cows was also beneficial in a situation where maize silage had to be purchased. The number of cows and area of grassland increased until intake capacity of the cow limited uptake of grass silage in winter. Total milk production at farm level increased to 1459 t FPCM/year, and labor income to €107,781. This corresponds to an increase of €123/unit change per cow per year.

Increasing longevity slightly increased energy and protein requirements per cow, explaining the increase in the amount of concentrates per cow per day during winter. The reduced replacement rate resulted in less young stock and lower manure production of the herd. The number of cows reached the level where maize silage had to be purchased. After increasing longevity, the number of cows on the farm was 150, total milk production 1380 t FPCM/year, and labor income €101,010/year. This corresponds to an increase of €77/unit change per cow per year.

Comparison of the reference scenarios of the less efficient and efficient farm (Chapter 5, Table 2), shows that the less efficient farm has fewer cows, a lower milk production per ha and higher production costs. Because of these higher production costs, no maize silage was purchased on the less efficient farm, while maize silage was purchased on the efficient farm. On the less efficient farm diets contained more concentrates and less maize silage per cow per day than on the efficient farm. Labor income was 22% lower on the less efficient farm than on the efficient farm.

On both the less efficient and the efficient farm, increasing milk yield resulted in a situation where maize land reduced in favor of grassland and (more) maize silage was purchased, to facilitate an increase in number of cows. Increasing longevity also increased the number of cows, but on the less efficient farm the increase stopped at the point where maize silage had to be purchased.

For the less efficient farm, economic values of milk yield and longevity were €123 and €77 per cow per year (i.e. method 1). For the efficient farm, these values were €122 and €82 per cow per year. The economic value of milk yield hardly depended on efficiency whereas the economic value of longevity increased with increase in farm efficiency. Thus, the economic importance of longevity relative to milk yield increased with increase in farm efficiency. In case of the less efficient farm, costs related to feeding the dairy cows determined a larger part of the total costs. An increase in longevity is mainly effective because it reduces the number of young stock. For the less efficient farm, costs related to young stock determine a smaller part of the total costs, reducing the importance of longevity relative to milk yield compared to the efficient farm.

Greenhouse gas emissions

Table 3 shows GHG emissions for the reference scenario and changes in emissions after increasing milk yield and longevity with one genetic standard deviation. Results in Table 3 are based on maximizing labor income (method 1).

For the less efficient farm the following results apply. In the reference scenario, total GHG emissions per t FPCM were 946 kg CO_2e without allocation, and 853 kg CO_2e based on economic allocation. Most important contributors were enteric fermentation (48%), production of concentrates (18%), manure (13%), and on-farm roughage production (13%).

Increasing milk yield reduced emissions per t FPCM due to dilution of emissions related to maintenance and young stock over more kg milk (i.e. fewer animals were needed to produce the same amount of milk). Furthermore, emissions changed because of changes in optimum farm management. After increasing milk yield, more roughage and less concentrates was fed per kg milk compared to the reference scenario. Because production of concentrates results in higher emissions than production of roughage, emissions related to feed production decreased. In total, GHG emissions per t FPCM decreased by 40 kg CO_2e using no allocation, and by 30 kg CO_2e using economic allocation. Based on economic allocation, the GHG value of milk yield was 279 kg CO_2e/unit change per cow per year.

Increasing longevity reduced emissions per t FPCM mainly by reducing the number of young stock needed for replacement (i.e. contributing to emissions from enteric fermentation, manure, and concentrate production). Furthermore, emissions changed because of changes in optimum

Table 3. GHG emissions for the reference scenario and the effect of increasing milk yield and longevity with one genetic standard deviation (in kg CO_2e/t FPCM) based on maximizing labor income.

	References		Milk yield		Longevity	
	L[1]	H[1]	L	H	L	H
Animal emissions						
Enteric CH_4 emission dairy cows	377	372	-5	-5	0	3
Enteric CH_4 emission young stock	73	73	-5	-5	-12	-12
Manure	126	118	-6	-5	-6	-6
On-farm feed production						
Grassland	80	67	5	6	-2	6
Maize land	40	37	-14	-14	-1	-13
Production of farm inputs						
Maize silage	0	24	19	18	0	17
Concentrates dairy cows	164	110	-30	-27	1	-23
Concentrates young stock	8	8	-1	-1	-1	-1
Synthetic fertilizer	55	51	-2	-2	-2	-2
Other inputs	23	23	-1	-1	0	0
Total emissions	946	882	-40	-36	-24	-32
Economic allocation[2]	853	796	-30	-27	-16	-23

[1] Columns indicated with an L apply to results of the less efficient farms, while columns indicated with an H applies to results of the efficient farm stemming from Van Middelaar et al. (2014).

[2] Economic allocation factors for milk were 0.90 for the reference scenario and 0.91 after increasing milk yield or longevity.

farm management. In total, GHG emissions per t FPCM reduced by 24 kg CO_2e using no allocation, and by 16 kg CO_2e using economic allocation. Based on economic allocation, the GHG value of longevity was 143 kg CO_2e/unit change per cow per year.

Compared to the efficient farm, the less efficient farm resulted in higher GHG emissions per t FPCM, i.e. 853 compared to 796 kg CO_2e/t FPCM (reference scenarios). Emissions from the production of purchased feed products, from on-farm feed production, and from enteric fermentation and manure management were higher on the less efficient farm than on the efficient farm. Results show that increasing farm efficiency via precision feeding and increasing roughage production per ha is an effective way to reduce GHG emissions from milk production.

Effects of increasing milk yield on GHG emissions on the less efficient farm are largely similar to effects on the efficient farm: on the less efficient farm emissions reduced by 3.5%, and on the efficient farm by 3.4%. Effects of increasing longevity on GHG emissions on the less efficient farm

Table 4. GHG values of milk yield and longevity (kg CO_2e/cow per year) for the less efficient and efficient farm based on maximizing labor income, or minimizing GHG emissions[1]

	Milk yield	Longevity	Ratio
Maximizing labor income			
Less efficient farm	279	143	1.94
Efficient farm	247	210	1.18
Minimizing GHG emissions			
Less efficient farm	538	563	0.95
Efficient farm	453	441	1.03

[1] Results are based on economic allocation.

are less pronounced than on the efficient farm: on the less efficient farm emissions reduced by 1.9%, whereas on the efficient farm emissions reduced by 2.9%.

For the less efficient farm, GHG values of milk yield and longevity were 279 and 143 kg CO_2e/unit change per cow per year (Table 4). For the efficient farm, these values were 247 and 210 kg CO_2e/unit change per cow per year. Thus, with an increase in farm efficiency, the GHG value of milk yield decreased, whereas that of longevity increased. As a result, the importance of milk yield relative to longevity is greater on the less efficient farm than on the efficient farm. These results show that GHG values of milk yield and longevity depend on farm efficiency when maximizing labor income.

3.2 Minimizing GHG emissions per kg FPCM (method 2)

Diets and farm plan

Table 5 shows the diets, farm plan, and farm outputs of the reference scenario and the scenarios in which milk yield and longevity were increased and farm management was optimized to minimize GHG emissions per kg milk (within the constraints set for labor income and milk production).

The reference scenario is the same as the reference scenario based on maximizing labor income. After increasing milk yield and subsequently minimizing GHG emissions per kg milk, the number of cows reduced from 145 to 137, and the amount of concentrates in the diets reduced. Labor income and total milk production were greater than in the reference scenario and consequently not restricting changes in farm management. This implies that the optimal solution within the feasible solution space was determined by other constraints. The number of cows, and hence total

Table 5. Diets, farm plan, and farm outputs of the less efficient farm, for the reference scenario and after increasing milk yield and longevity with one genetic standard deviation, based on minimizing GHG emissions.

		Reference	Milk yield	Longevity
Diet dairy cows - summer period (kg DM/cow/day)				
Grass (grazed)		12.0	12.0	12.0
Maize silage		5.1	7.7	7.0
Concentrates - medium protein		3.7	-	-
- high protein		-	2.4	2.0
Diet is restricted by[1]		E,R	E,R	E,R
Diet dairy cows - winter period (kg DM/cow/day)				
Grass silage		4.9	5.8	5.9
Maize silage		5.3	8.3	8.6
Concentrates - medium protein		7.1	1.8	-
- high protein		-	2.7	3.4
Diet is restricted by [1]		E,R	E,R,I	E,R,I
Farm plan				
Dairy cows	n	145	137	148
Young stock	unit [2]	44	42	38
Grassland 225 kg N/ha	ha	60.6	59.5	62.8
Maize land	ha	24.4	25.5	22.2
Synthetic fertilizer	kg N/ha	100	101	102
	kg P_2O_5/ha	-	-	-
Purchased maize silage	t DM	-	109	169
Purchased concentrates	t DM	310	192	163
Manure application is restricted by [3]		P	P	P
Farm outputs				
Milk	t FPCM	1336	1360	1362
Dairy cows	n	39	37	33
Young stock > 12 months	n	3.4	3.2	2.9
Young stock < 12 months	n	1.8	1.7	1.5
Calves	n	101	95	110
Labor income	€	89,885	106,363	99,288

[1] The diet can be restricted by: E = energy requirements; R = rumen degradable protein balance; T = true protein digested in the small intestine; I = intake capacity.

[2] One unit includes 1 animal < 12 months and 0.96 animal > 12 months.

[3] The intensity of the farm is restricted by the possibility to apply manure. Manure application can be restricted by: tN = total mineral N; aN = N from animal manure; P = total P_2O_5.

milk production, was constrained by the amount of grass and grass silage produced (i.e. all grass was consumed) in combination with the maximum intake capacity of the cow. The area of grassland was 59.5 ha, which is exactly 70 % of the total ha available on farm, i.e. the minimum area to allow application of maximal 250 kg N/ha from animal manure instead of maximal 170 kg N/ha. The number of cows and the amount of grass (silage) could have been reduced by reducing N fertilization on grassland. Due to a minimum requirement for total milk production (thus a minimum number of cows) and a minimum requirement for RDPB per cow, however, reducing N fertilization would have decreased grass production to a level that would have required an increase in the amount of concentrates per cow. Although concentrates result in a lower enteric CH_4 emission than grass and grass silage, emissions during production are higher. Reducing the amount of concentrates in the diets, therefore, was more beneficial in terms of GHG emissions per kg milk than reducing N fertilization on grassland. After increasing milk yield, and subsequently minimizing GHG emissions per kg milk, total milk production at farm level increased to 1360 t FPCM/year, and labor income to €106,363/year.

After increasing longevity, the number of cows increased from 145 to 148, and the amount of concentrates in the diets reduced. Again, labor income and total milk production were not restricting. With total milk production as a minimum constraint, the number of cows and the area of grassland increased until the maximum intake capacity of the cow was met. The area of grassland increased to facilitate the increase in the number of cows, because P_2O_5 application standards were limiting and application standards are higher for grassland than for maize land. The number of cows and area of grassland increased as long as the increase in grass and grass silage was fully consumed (i.e. intake capacity was restricting). This balance was reached at 148 cows. After increasing longevity, total milk production increased to 1362 t FPCM/year, and labor income to €99,288/year.

Comparing results after increasing milk yield and longevity for the less efficient farm with results for the efficient farm (Chapter 5, Table 4) shows three major differences. First, N fertilization on grassland was not reduced on the less efficient farm, whereas on the efficient farm it was. Due to higher RDPB requirements on the less efficient farm, reducing N fertilization on grassland, resulting in a lower RBPB content per kg DM grass and grass silage, was not beneficial for reducing GHG emissions per kg milk. Second, on both farms and for both traits, the amount of concentrates in the diets was reduced and the amount of roughage was increased. On the less efficient farm, however, the amount of concentrates per cow per day was larger than on the efficient farm, due to higher RDPB requirements. Third, on both farms and for both traits the number of cows was increased until P_2O_5 application standards in combination with the maximum intake capacity of the cow was restricting. On the less efficient farm, the final number of cows was lower than on the efficient farm. This was caused by a higher P content of manure on the less efficient farm because of higher RDPB requirements and a subsequent higher P content in the diet.

Greenhouse gas emissions

Table 6 shows GHG emissions of the reference scenario and changes in emissions after increasing milk yield and longevity. Farm management was optimized to minimizing GHG emissions per kg milk. After increasing milk yield per cow on the less efficient farm, emissions decreased due to dilution of emissions related to maintenance over more kg milk, and due to changes in optimal farm management. Emissions from concentrate production decreased, while emissions from maize silage increased. Overall, emissions per t FPCM decreased by 71 kg CO_2e using no allocation, and by 59 kg CO_2e using economic allocation. Based on economic allocation, the GHG value of milk yield is 538 kg CO_2e/unit change per cow per year.

After increasing longevity, emissions related to young stock (enteric fermentation, manure, and concentrate production) decreased because of a reduction in young stock (i.e. reduced

Table 6. GHG emissions for the reference scenario and the effect of increasing milk yield and longevity with one genetic standard deviation (in kg CO_2e/t FPCM) based on minimizing GHG emissions[1].

	References		Milk yield		Longevity	
	L[1]	H[1]	L	H	L	H
Animal emissions						
Enteric CH_4 emission dairy cows	377	372	-8	-2	3	11
Enteric CH_4 emission young stock	73	73	-5	-5	-12	-12
Manure	126	118	-5	-5	-5	-5
On-farm feed production						
Grassland	80	67	-2	-1	3	8
Maize land	40	37	0	-5	-5	-20
Production of farm inputs						
Maize silage	0	24	14	15	22	32
Concentrates dairy cows	164	110	-64	-48	-79	-62
Concentrates young stock	8	8	-1	-1	-1	-1
Synthetic fertilizer	55	51	0	-8	0	-8
Other inputs	23	23	0	0	0	0
Total emissions	946	882	-71	-60	-74	-59
Economic allocation[2]	853	796	-59	-49	-61	-48

[1] Columns indicated with an L apply to results of the less efficient farms, while columns indicated with an H apply to results of the efficient farm stemming from Van Middelaar et al. (2014).

[2] Economic allocation factors for milk were 0.90 for the reference scenario and 0.91 after increasing milk yield or longevity.

replacement rate). In addition, emissions decreased due to changes in optimal farm management. Overall, emissions per t FPCM decreased by 74 kg CO_2e using no allocation, and by 61 kg CO_2e using economic allocation. Based on economic allocation, the GHG value of longevity is 563 kg CO_2e/unit change per cow per year.

Differences between results of the less efficient and efficient farm can be explained mainly by the differences discussed in the paragraph on diets and farm plan. After increasing milk yield, the reduction in emissions from concentrates production was greater on the less efficient farm than on the efficient farm, whereas the reduction in emissions related to on-farm roughage production was less. The reduction in enteric CH_4 emission is greater on the less efficient farm than on the efficient farm. This results from greater RDPB requirements, and hence, a greater amount of concentrates in the diets in case of the less efficient farm. After increasing longevity, a similar pattern is shown. Additional differences in emissions related to on-farm feed production result from differences in an increase in grassland at the expense of maize land, i.e. on the efficient farm a larger part of maize land was changed into grassland.

For the less efficient farm, GHG values of milk yield and longevity were 538 and 563 kg CO_2e/unit change per cow per year (Table 4). For the efficient farm, these values were 453 and 441 kg CO_2e/unit change per cow per year. Thus, with a decrease in farm efficiency, GHG values of both milk yield and longevity increased. The ratios between GHG values of both traits, however, are comparable on both farms: on the less efficient farm and on the efficient farm, milk yield and longevity are about equally important for reducing GHG emissions.

4 Conclusions

The impact of feed-related farm characteristics on GHG values of genetic traits was evaluated by comparing GHG values of milk yield and longevity for a less efficient farm with those for an efficient farm. When optimizing farm management based on maximizing labor income, the GHG value of milk yield relative to the value of longevity decreased with increase in farm efficiency. Thus, GHG values of milk yield and longevity depend on farm efficiency when maximizing labor income. The reason is that optimizing farm management based on maximizing labor income does not always result in an optimal situation for reducing GHG emissions.

When optimizing farm management based on minimizing GHG emissions per kg milk, GHG values of both traits decreased with increase in farm efficiency, but the ratio between values hardly changed: both traits were equally important on each level of efficiency. On both farms, GHG values based on minimizing GHG emissions were at least twice as great as GHG values based on maximizing labor income.

Chapter 7

General discussion

1 Introduction

Reducing greenhouse gas (GHG) emissions from dairy production has become an imperative study object. Important areas of interest to reduce GHG emissions per kg milk include feeding strategies to reduce emissions from enteric fermentation and feed production, and breeding strategies to improve animal productivity. Most studies that evaluate the potential of feeding and breeding strategies to reduce GHG emissions, however, do not account for emissions other than enteric methane (CH_4), do not account for changes in farm management to adapt the farm optimally to the particular strategy, or do not account for consequential effects in other parts of the milk-production chain.

The two objectives of this thesis were to develop an integrated method to evaluate strategies to reduce GHG emissions from dairy production at the chain level, and to evaluate feeding and breeding strategies using this integrated method. This chapter, therefore, starts with a discussion on the relevance and methodological challenges of integrated modeling. Subsequently, the potential of specific feeding and breeding strategies to reduce GHG emissions from dairy production is presented and placed in a wider context. Finally, practical implications for reducing GHG emissions are discussed, and an overview of the conclusions from this thesis is given.

2 Integrated modeling of strategies

2.1 Relevance of integrated modeling

The relevance of integrated modeling of strategies to reduce GHG emissions was demonstrated in Chapter 3 by evaluating the impact of increasing maize silage at the expense of grass and grass silage in a dairy cow's diet at three levels: animal, farm, and chain levels. In this chapter, we combined a whole-farm optimization model with a life cycle approach and a mechanistic model to predict enteric CH_4 production.

Whole-farm models are developed to describe and quantify flows of materials and nutrients within a farming system, and have been used frequently to explore economic and environmental consequences of policy changes or innovations in dairy systems (Van Calker et al., 2004; Fiorelli et al., 2008; Crosson et al., 2011). Feeding and breeding strategies to reduce GHG emissions from dairy production are examples of innovations that can change nutrient (e.g. carbon and nitrogen) flows within the system, and farm management. These changes in nutrient flows and farm management can change GHG emissions, such as emissions of CH_4 and nitrous oxide (N_2O) from manure management, and emissions of carbon dioxide (CO_2) and N_2O from on-farm feed production. Evaluation of strategies to reduce GHG emissions, therefore, requires a method that accounts for the interaction between farm components and the interrelated consequences on all GHG emissions (Crosson et al., 2011; Del Prado et al., 2013).

Whole-farm models have been used frequently to evaluate the impact of strategies to reduce GHG emissions (e.g. Weiske et al., 2006; Del Prado et al., 2010; Beukes et al., 2011; Misselbrook et al., 2013). Most studies, however, used a whole-farm simulation model rather than an optimization model. In a simulation model, the reference farm and solution options are an input of the model. A simulation model, therefore, is suitable to evaluate the impact of a strategy in a specific situation, e.g. a particular farm for which the management variables and solution options are known. To draw a more general conclusion about the impact of a strategy, an optimization model is required. An optimization model includes a guiding principle that guarantees an optimal solution before and after implementing the strategy. Solutions are based on optimization of farm management to maximize a given objective rather than on arbitrary choices. Because earning a decent income is the main objective of most farmers, economic optimization is most suitable. Throughout this thesis, it has been shown that most strategies are likely to change farm management. This confirms the importance of correct evaluation of these changes. In Chapter 3, for example, increasing maize silage in the dairy cow's diet at the expense of grass and grass silage increased the amount of purchased concentrates, while the amount of purchased maize silage decreased. In Chapters 5 and 6, increasing milk yield per cow decreased the amount of purchased concentrates and increased the area of grassland, while the area of maize land decreased. So far, studies that used a whole-farm optimization model to evaluate strategies to reduce GHG emissions are limited (Van Calker et al., 2004; Gibbons et al., 2006; Adler et al., 2013).

Whole-farm models (both simulation and optimization) generally use empirical relations to estimate the emission of enteric CH_4. A method that is often used is IPCC Tier 2 (Del Prado et al., 2013). IPCC Tier 2 estimates the emission of enteric CH_4 based on animal dry matter (DM) intake or metabolizable energy intake, while assuming a fixed CH_4 conversion factor independent of the diet. For dairy cattle, for example, CH_4 energy output is assumed to be 6.5% of gross energy intake. In this thesis, we used a mechanistic model to estimate the emission of enteric CH_4 (Dijkstra et al., 1992; Bannink et al., 2006). Mechanistic models are more accurate to evaluate the impact of dietary changes on enteric CH_4 production than empirical models (e.g. Benchaar et al., 1998; Kebreab et al., 2006b). In Chapter 3, for example, increasing maize silage by 1 kg DM/cow per day at the expense of grass and grass silage, reduced the emission of enteric CH_4 by 3.2%, i.e. 13 kg CO_2 equivalents (CO_2e)/ton fat-and-protein-corrected milk (FPCM). Comparing these results with those using the calculation methods of IPCC Tier 2 shows the importance of mechanistic modeling to evaluate the impact of dietary changes. Based on IPCC Tier 2 (i.e. gross energy content of feed assumed to be 18.45 MJ/kg DM), the very same strategy resulted in a reduction of 0.3%, i.e. 1 kg CO_2e/ton FPCM. At the chain level, the reduction is 3 kg CO_2e/ton FPCM, which is one fifth the reduction of 14 kg CO_2e/t FPCM that was presented in Chapter 3. So far, one other study combined a whole-farm model with mechanistic modeling of enteric CH_4. Beukes et al. (2011) evaluated the impact of several strategies (e.g. improved reproductive performance of the herd, reduced nitrogen fertilization on grassland) on GHG emissions from dairy farms in New Zealand. Beukes et al. (2011), however, used a simulation model and GHG calculations that did not include the entire production chain (i.e. from production of farm inputs up to the farm gate).

Regarding the environmental impact, whole-farm models are generally restricted to the farm level. A strategy that reduces GHG emissions at the farm level, however, might increase GHG emissions related to production of farm inputs. Such a strategy offers little potential to reduce the impact of dairy production on GHG emissions. Throughout this thesis, the importance of including emissions related to the production of farm inputs has been confirmed. In Chapter 4, for example, we showed that supplementing the diet with a composed linseed product reduced GHG emissions at the farm level by almost 7%, whereas the reduction at chain level was only 1%. Including emissions related to the production of farm inputs is, therefore, important to understand the full impact of a strategy on GHG emissions. No other study combined a whole-farm optimization model with GHG calculations of farm inputs, although several studies combined a simulation model with GHG calculations of some of the major farm inputs (e.g. Weiske et al., 2006; Del Prado et al., 2010; Beukes et al., 2011). Most of these studies, however, did not include the entire production chain.

A method that can be used to account for all GHG emissions along the dairy production chain is life cycle assessment (LCA). Several studies used LCA to compare the environmental impact of different dairy farms or to identify improvement options (Kristensen et al., 2011; Yan et al., 2013; Meul et al., 2014). Some studies used LCA to evaluate strategies to reduce GHG emissions along the production chain (Williams et al., 2013; Schader et al., 2013). These studies, however, did not account for changes in farm management to adapt the farm optimally to the particular strategy. To verify if a strategy actually offers potential to reduce GHG emissions, changes in farm management have to be taken into account. As a single instrument, LCA lacks the ability to account for these changes and, therefore, should not be used to evaluate strategies to reduce emissions.

By combining a whole-farm model based on economic optimization with a mechanistic model to predict enteric CH_4 production and LCA, this thesis provides an effective method to evaluate strategies to reduce GHG emissions from dairy production. In addition, the method can be used to evaluate the economic consequences of a strategy. Insight into economic consequences is required to determine if a strategy has potential to be used on a commercial farm (Hristov et al., 2013b).

By combining economic consequences with impact on GHG emissions, the cost-effectiveness of a strategy can be calculated (Chapter 4). Several studies have calculated the cost-effectiveness of strategies to reduce GHG emissions from dairy production (Del Prado et al., 2010; Vellinga et al., 2011; Adler et al., 2013). Most of these studies, however, did not use an economic optimization model. Economic optimization of farm management, before and after implementing a strategy, is required to prevent under- or over-estimation of the economic consequences (Groen et al., 1997; Adler et al., 2013). The integrated method, furthermore, can be used to calculate the value of a genetic trait in the breeding goal, i.e. the impact of one genetic standard deviation improvement of a trait on labor income and GHG emissions. Current breeding goals are generally based on economic values. In the future, however, the importance of a genetic trait to contribute to a reduction in GHG emissions might increase.

2.2 Methodological challenges of integrated modeling

Accuracy of methods and data

Evaluation of strategies to reduce GHG emissions requires accurate methods and data to calculate emissions from each process along the chain (Lengers et al., 2013). Emissions from enteric fermentation and feed production are the two most important contributors to total GHG emissions from dairy production (Chapter 1). To evaluate feeding and breeding strategies, correct assessment of the impact on enteric CH_4 and on GHG emissions from feed production are most relevant.

Several studies have evaluated the accuracy of methods to estimate enteric CH_4 production (Benchaar et al., 1998; Kebreab et al., 2006b), but information on the accuracy of methods and data to calculate GHG emissions from feed production is limited. Chapter 2, therefore, explored the impact of differences in methods and data on GHG emissions per kg feed. Results showed that differences in methods to calculate nitrate leaching, which is required to estimate indirect N_2O emissions, and differences in methods to calculate emissions from changes in land use, significantly affect results. In addition, GHG calculations were not robust to assumptions on crop yield per hectare and use of synthetic fertilizer (i.e. nitrogen). Accuracy of GHG calculations can be increased by using country- and site-specific data (IPCC, 2006). In case of purchased feed products, however, the use of site-specific data is often hampered by lack of insight into production circumstances or into the exact origin of the product.

To limit the impact of methodological choices on the evaluation of strategies, we used one generally accepted method to calculate emissions from purchased feed products (i.e. IPCC Tier 1 according to Vellinga et al., 2013). To account for the impact of temporal and spatial variation in production circumstances, and to account for differences in methods to calculate emissions from land use change, a sensitivity analysis was performed for each strategy. A sensitivity analysis provides insight into the robustness of results to assumptions in methods and data (Zehetmeier et al., 2013). Consistency in methods and evaluation of the robustness of results are important aspects when evaluating the impact of a strategy.

Modeling soil carbon fluxes

Changes in crop management (e.g. tillage system, manure application) and land use (e.g. conversion of grassland into maize land) can affect the amount of carbon and nitrogen in the soil and contribute to either CO_2 and N_2O emissions, or to CO_2 sequestration (Chapter 2). Soil CO_2 sequestration in agricultural land, particularly grassland, is an important mechanism to reduce GHG emissions (Soussana et al., 2010). When crop management and land use remain unchanged for a period of time, soil carbon and nitrogen stocks seek a new equilibrium. When this new equilibrium is reached, emissions (sequestration) no longer occur. Emissions from changes in crop management and land use (i.e. non-recurrent), therefore, cannot be compared with changes in annual (i.e. recurrent) emissions, such as enteric CH_4 emission, and emissions from manure management. The impact of a strategy on non-recurrent emissions has to be weighed against its impact on recurrent emissions. By calculating the carbon payback time of a strategy that increases non-recurrent emissions and decreases recurrent emissions, for example, insight is gained into the number of years that are needed before non-recurrent emissions are compensated (Chapter 3).

Handling a multiple-output situation

Dairy cattle produce not only milk but also meat and manure. In such a multiple-output situation, GHG emissions of the dairy system have to be allocated to the various outputs. Most studies allocate GHG emissions from dairy production to milk, meat, and manure based on economic allocation, i.e. on their relative economic value (De Vries and De Boer, 2010). Economic allocation implies, for example, that emissions related to production of meat from surplus calves and to processing of culled cows that take place outside the dairy farm are excluded from the analysis. Instead of economic allocation, system expansion can be used. System expansion implies, for example, that GHG emissions related to production of meat from surplus calves and to processing of culled cows are attributed to dairy production, and thus to the milk. At the same time, meat from surplus calves and culled cows is recognized as a valuable co-product that can substitute for other meat products, such as pork, chicken, or beef from suckler cows. For system expansion, emissions related to the production of meat products that are assumed to be replaced with meat from surplus calves and culled cows are deducted from emissions related to dairy production, and thus from the milk.

In this thesis, the dairy production system yielded only milk and meat and not manure, because we assumed that all manure was used on the farm. The choice of method for handling a multiple-output situation was important, therefore, especially when evaluating a strategy that affected the ratio of milk to meat production. An example of a strategy that influences the ratio of milk to meat production is increasing milk yield per cow. For economic allocation, increasing milk yield per cow reduces GHG emissions per kg milk, because of dilution of emissions related to maintenance over more kg milk (Capper et al., 2011). For system expansion, however, the impact of increasing milk yield per cow depends on the type of product that is assumed to be replaced with meat from surplus calves and culled cows. Increasing milk yield per cow results in decreasing the amount of meat produced per kg milk. Assuming that the total demand for milk and meat remains unchanged, this implies that production of pork, chicken, or beef must increase. If increasing milk yield results in an increase in the production of beef from suckler cows, i.e. a product with a high impact on GHG emissions, the impact of an increase in milk yield on global GHG emissions is less positive than results based on economic allocation indicate (Chapter 5). Differences between results based on economic allocation and results based on system expansion emphasize the importance of the choice of the method for handling a multiple-output situation when evaluating strategies that affect the ratio of milk to meat production.

Sustainability aspects

In this thesis, we considered the environmental impact of strategies related to GHG emissions from dairy production. Dairy production, however, has an impact on the environment in other ways, such as eutrophication, acidification, and depletion of fossil energy and phosphorus sources (Guerci et al., 2013). Dairy production, furthermore, can contribute to ecosystem services, e.g. by conserving biodiversity and nature, and by maintaining cultural landscapes. To evaluate the impact of strategies on other environmental issues and on ecosystem services, and to evaluate trade-offs, the integrated method presented in this thesis can be extended. In addition to environmental and economic aspects, animal welfare is an important sustainability aspect that needs to be taken into account when selecting for strategies to reduce GHG emissions.

3 Feeding and breeding strategies to reduce GHG emissions

3.1 Feeding strategies

A summary of the impact of feeding and breeding strategies on GHG emissions and labor income as reported in this thesis is shown in Table 1. Feeding strategies can be realized at the short term and affect GHG emissions right away. The impact of feeding strategies was evaluated for an typical Dutch dairy farm, under the current milk quota system.

Each feeding strategy examined reduced GHG emissions along the chain. More detailed results (Chapters 3 and 4) showed that, for most strategies, changes in emissions from feed production partly off-set the reduction in enteric CH_4 emission. Supplementing diets with nitrate resulted in the largest reduction at the chain level (4%), whereas the impact of other feeding strategies was lower (about 1%). Emissions related to changes in land use are not included in Table 1, because these emissions are non-recurrent, whereas all other results relate to recurrent (annual) emissions. Non-recurrent emissions related to on-farm land use change, however, have an important negative impact for replacing grass and grass silage with maize silage (Chapter 3).

Results showed that each feeding strategy reduced labor income. Reduction in labor income was smallest for the strategy that implies a reduction in maturity of grass and grass silage. Combining the impact on labor income with the impact on GHG emissions showed that this latter strategy is most cost-effective, and, therefore, offers most potential to be implemented on commercial farms (Chapter 4).

Table 1. Summary of the impact of feeding (upper half) and breeding[1] (lower half) strategies on GHG emissions[2] (kg CO_2e/t FPCM) and labor income (€/year) as reported in this thesis.

Thesis section	Strategy	Reference situation		Effect of the strategy *maximizing income*				Effect of the strategy[3] *minimizing GHGs*			
		GHGs	Income	GHGs	(%)	Income	(%)	GHGs	(%)	Income	(%)
Current farm											
Chapter 3	Repl. grass (silage) with maize silage	909	34,969	-14	(-2)	-2,305	(-7)				
Chapter 4	Supplementation of extruded linseed	840	42,605	-9	(-1)	-16,041	(-38)				
	Supplementation of dietary nitrate			-32	(-4)	-5,463	(-13)				
	Reducing maturity stage of grass (silage)			-11	(-1)	-463	(-1)				
Future farm[4]											
Chapter 5	Increasing milk yield (efficient farm)	796	115,050	-27	(-3)	20,427	(+18)	-49	(-6)	12,251	(+11)
	Increasing longevity (efficient farm)			-23	(-3)	13,715	(+12)	-48	(-6)	5,378	(+5)
Chapter 6	Increasing milk yield (less efficient farm)	853	89,885	-30	(-4)	17,896	(+20)	-59	(-7)	16,478	(+18)
	Increasing longevity (less efficient farm)			-16	(-2)	11,125	(+12)	-61	(-7)	9,403	(+10)

[1] Results show the impact of one genetic standard deviation improvement in milk yield and longevity.
[2] Based on economic allocation.
[3] Impact of increasing milk yield and longevity was also evaluated based on the objective to minimize GHG emissions per kg milk.
[4] Impact of breeding strategies were calculated for a farm in 2020. Results of feeding and breeding strategies are not comparable.

Throughout this thesis, several other potential strategies to reduce GHG emissions have been identified, i.e. supplementing diets with urea, reducing or eliminating safety margins for rumen degradable protein and for true protein digested in the small intestine, increasing animal and plant productivity, and reducing the amount of concentrates in the diet. Strategies were identified by comparing reference situations across chapters and by comparing the two methods used in Chapters 5 and 6. To place the impact of the feeding strategies evaluated in Chapters 3 and 4 in a wider context, the following paragraphs focus on comparison of results across chapters.

In Chapter 3, total GHG emissions of the reference situation were estimated to be 909 kg CO_2e/ton FPCM. In Chapter 4, total GHG emissions of the reference situation were estimated to be 840 kg CO_2e/ton FPCM, i.e. about 8% less than emissions reported in Chapter 3. Differences in emissions between the reference situations can be explained mainly by three aspects. First, in Chapter 3 annual yield of maize silage was assumed to be 13.3 ton DM/ha, whereas in Chapter 4 it was 15.5 ton DM/ha. Second, in Chapter 3 safety margins for rumen degradable protein were set at 200 gram/cow per day, whereas in Chapter 4 they were 100 gram/cow per day. Third, Chapter 3 did not include the option to feed urea, whereas Chapter 4 did. About 50% of the difference in GHG emissions between the two reference situations was explained by the difference in maize yield per ha, and about 50% by reducing safety margins for rumen degradable protein and including the option to feed urea (data not presented). As a result of reducing safety margins and including the option to feed urea, diets in Chapter 4 contained more maize silage and less concentrates than diets in Chapter 3. Compared to concentrates, maize silage results in less emissions during production and enteric fermentation.

Increasing maize silage yield per ha, reducing safety margins for rumen degradable protein, and supplementing diets with urea offer potential to decrease GHG emissions per kg milk, without compromising labor income. Feeding urea can be an economically viable strategy to reduce GHG emissions because the costs of urea are low compared with other protein sources. Feeding high levels of urea, however, can negatively affect feed intake, production traits, and animal health (Brito and Broderick, 2007). Amounts of supplemental urea in this study (i.e. 20 g/cow per day in summer and 60 g/cow per day in winter; Chapter 4), however, were within the recommended maximum of 135 g/cow per day (Kertz, 2010).

Comparing results of Chapters 3 and 6 provide insight into the potential impact of an increase in animal and plant productivity on GHG emissions per kg milk. In Chapter 6, we simulated a less efficient farm in the technical and institutional settings of 2020. Total GHG emissions of the reference situation in Chapter 6 were 853 kg CO_2e/ton FPCM, which was about 6% less than emissions reported in Chapter 3 (909 kg CO_2e/ton FPCM). Differences between Chapters 3 and 6 mainly relate to differences in replacement rate and milk yield per cow, and a difference in crop yield per ha. In Chapters 3 and 4, replacement rate was assumed to be 29% and milk yield was assumed to be 7968 kg/cow per year. In Chapters 5 and 6, however, replacement rate was assumed to be 27% and milk yield was assumed to be 8758 kg/cow per year. Predicted yield per ha for purchased and on-farm feed products were greater in Chapter 6 than in Chapter 3.

In Chapter 5, we simulated an efficient farm in the setting of 2020. Comparing results of Chapters 5 and 6, therefore, show the impact of an increase in farm efficiency. Total GHG emissions of the reference situation in Chapter 5 were 796 kg CO_2e/ton FPCM, which was about 7% less than emissions reported in Chapter 6 (853 kg CO_2e/ton FPCM). Chapter 5 did not include safety margins for rumen degradable protein and true protein digested in the small intestine, whereas in Chapter 6 safety margins were set at respectively 200 and 100 g/cow per day. In Chapter 5, furthermore, grass and maize yields per hectare were about 5% greater than yields reported in Chapter 6.

A final strategy that was identified to reduce GHG emissions was reducing the amount of concentrates in the diet. The potential of this strategy was based on a comparison of results of the two methods used in Chapters 5 and 6 to evaluate the impact of genetic improvement on GHG emission (Table 1). The first method was based on maximizing labor income, whereas the second was based on minimizing GHG emissions per kg milk. Differences in results between the two methods are explained by differences in farm management and feeding strategy. The main difference was that for the second method, the amount of concentrates in the diets was reduced, whereas for the first it was not. Reducing the amount of concentrates (in combination with a few other changes) reduced emissions by about 3% to 5%. Production of concentrates resulted in greater GHG emissions than production of roughage, because yields per ha averaged less for concentrate ingredients than for roughage (i.e. grass and maize silage), and because concentrate ingredients were dried and processed, which contributed to greater GHG emissions. Emissions from concentrate production, however, depend on concentrate composition (i.e. emissions vary among ingredients), and the option to change the composition of concentrates was not included in the model. Overall, selecting for feed products with a low impact on GHG emissions can contribute to reducing GHG emissions from dairy production. Reducing the amount of concentrates over roughage might be an option to achieve this.

3.2 Breeding strategies

Breeding strategies (e.g. including the expected impact on GHG emissions in selection decisions), affect GHG emissions in the long term. To evaluate the impact of breeding strategies, therefore, the model farm was adapted to future production circumstances without a milk quota. Differences in labor income (Table 1) between the reference situations of Chapters 3 and 4 (feeding strategies) with those of Chapters 5 and 6 (breeding strategies) are explained mainly by an increase in farm size, greater forage production per ha, and change in prices. Because only prices of important in- and outputs were changed, and because price predictions contain uncertainty, the impact of breeding strategies on labor income should be judged on their relative impact.

Results of breeding strategies (Table 1) represent the impact of one genetic standard deviation improvement in milk yield and longevity, while other traits are kept constant. In practice, genetic selection is based on many traits simultaneously, and realised selection responses depend on the selection intensity for the trait of interest. Determining the impact of a multi-trait selection strategy requires knowledge of genetic parameters (i.e. heritability, genetic correlation) and the values of individual traits in the breeding goal. Results of Chapters 5 and 6, therefore, provide an important first step towards a better understanding of the potential of breeding to reduce GHG emissions from dairy production, and whether or not including the impact of genetic traits on GHG emissions will change selection decisions. Due to differences between feeding and breeding strategies under study, results of the strategies cannot be compared directly.

In Chapters 5 and 6, two methods were used to calculate GHG values of genetic traits. The GHG value of a genetic trait represents the impact of one unit change (i.e. one genetic standard deviation improvement) on GHG emissions at chain level while keeping other traits constant, and is expressed in kg CO_2e/unit change per cow per year (i.e. similar to economic values). For interpretation and comparison of results, Table 1 presents the impact of one unit change of a genetic trait in kg CO_2e/ton FPCM. The ratio between results expressed in CO_2e/ton FPCM are comparable to the ratio between results expressed in CO_2e/cow per year (i.e. the GHG values).

The first method that was used to calculate GHG values optimized farm management before and after a change in genetic trait by maximizing labor income (i.e. the same method that is used to calculate economic values). The second method optimized farm management after a change in genetic trait by minimizing GHG emissions per kg milk, while maintaining labor income and total milk production of the herd at least at the level before a change in genetic trait. In Chapter 5, GHG values of milk yield and longevity were calculated for an efficient farm, whereas in Chapter 6 they were calculated for a less efficient farm. Differences between the efficient and less efficient farm relate to differences in safety margins for protein and in grass and maize silage yields per ha.

Comparing the impact of one genetic standard deviation improvement in milk yield and longevity on GHG emissions per ton FPCM shows that results varied between methods and farms (Table 1). When the objective was to maximize labor income, the importance of milk yield to reduce GHG emissions was greater than that of longevity. When the objective was to minimize GHG emissions, however, both traits were equally important. Results indicate that the importance of longevity relative to milk yield increases as importance of reducing GHG emissions increases. Milk yield has a greater economic value than longevity, and, therefore a greater importance in the current breeding goal. In a situation where GHG emissions have a price (e.g. via subsidies or carbon taxes), therefore, milk yield remains more important than longevity until a specific price level. Including GHG emissions costs into economic values of traits for beef cattle had no effect on the breeding goal (Åby et al., 2013). Åby et al. (2013), however, only included GHG emissions related to enteric fermentation and manure management, and set three prices per ton CO_2e: €35, €70, and €90. An increase in price moderately affected the relative economic values of genetic traits (Åby et al., 2013). Conducting a similar study on genetic traits for dairy cattle can contribute to a better understanding of the impact of carbon taxes on future breeding goals.

Comparing results of the efficient and less efficient farm shows that the impact of one genetic standard deviation improvement in milk yield and longevity on GHG emissions depends on farm efficiency. Absolute reductions in GHG emissions were greater on the less efficient farm than on the efficient farm, except for an increase in longevity when based on maximizing labor income. When the objective was to maximize labor income, the importance of milk yield relative to longevity increased with decrease in farm efficiency. When the objective was to minimize GHG emissions, however, both traits were equally important on each level of efficiency. Results show that the impact of efficiency on the relative importance of genetic traits to reduce GHG emissions can be large, when using a method based on maximizing labor income. The reason is that optimizing farm management based on maximizing labor income does not always result in an optimal situation for reducing GHG emissions.

3.3 Applicability to other countries

The integrated method presented in this thesis can be used by other countries that want to gain insight into strategies to reduce GHG emissions. Feeding and breeding strategies were evaluated for the case-study of a typical Dutch dairy farm on sandy soil. Results, therefore, are specific for the Dutch situation. Overall conclusions, however, can be used as an indicator in other countries for which the dairy sector and climatologically conditions are comparable to the Dutch situation. A specific element of the Dutch dairy sector, for example, is the derogation regulation. The derogation regulation applies to a few countries in the EU that have a large proportion of their area in grassland (EU, 2010). The regulation prescribes that farms with at least 70% of their area in grassland may apply 250 kg N/ha originating from animal manure, instead of the 170 kg N/ha that is prescribed for farms with less than 70% in grassland. Reducing the area of grassland to less than 70%, therefore, has important economic consequences for a Dutch dairy farm, because it implies that manure has to be removed from the farm, which can be costly. This thesis showed that the derogation regulation influences the impact of a strategy on farm management and, therefore, the strategy's impact on GHG emissions (e.g. Chapter 3). Countries in the EU that do not have a derogation regulation might find different results and might, therefore, use different strategies to reduce GHG emissions than countries that do have a derogation regulation.

4 Practical implications

Identification of strategies to reduce GHG emissions is a first step towards reducing the impact of dairy production in practice. This thesis emphasized the importance of an integrated approach to evaluate strategies, and to take into account the economic consequences at the farm level. In addition to the feeding and breeding strategies that were evaluated in this thesis, other strategies (e.g. using different feeding strategies, reducing nitrogen fertilization on grassland, using manure processing) can be evaluated by using a similar approach. To evaluate breeding strategies, GHG values of genetic traits, other than milk yield and longevity, need to be estimated (e.g. feed efficiency, calving interval). Based on economic and GHG values of individual traits, and on current estimates of genetic parameters, a multi-trait selection strategy can be defined. Integrated modeling can then be used to compare the impact of multi-trait selection strategies for determining the breeding goal.

To contribute to a reduction in GHG emissions in practice, strategies need to be implemented on commercial farms. The Dutch dairy sector made a national commitment to reduce its levels of emissions by 30% in 2020, relative to 1990 levels (Clean and Efficient Agricultural Sectors Agreement, 2011). Strategies presented in this thesis reduced GHG emissions by 1% to 8%. Results indicate that a combination of different strategies (e.g. increasing plant and animal productivity, reducing safety margins for rumen degradable protein, and supplementing diets with urea or nitrate) is required to substantially reduce GHG emissions from dairy production. Other studies came to the same conclusion (e.g. Beukes et al., 2011).

To implement strategies successfully on commercial farms, it is important to acknowledge that each farmer has a management strategy and that options to reduce GHG emissions can vary between farms (Meul et al., 2014). In Chapter 3, for example, it was shown that increasing maize silage in a dairy cow's diet, at the expense of grass and grass silage, is not an option for Dutch dairy farmers who comply almost or entirely with the 70% grassland requirement of the derogation regulation. The impact of a strategy, furthermore, can vary between farms, because changes in farm management, hence, changes in GHG emissions, depend on the initial diet and farm management. To identify farm specific strategies, a benchmark or decision-support tool is required that provides insight into the GHG emissions of a farm and into the impact of options for improvement (Meul et al., 2014). Farm advisors can play an important role in identifying strategies. To avoid pollution swapping or carbon leakage, strategies should be selected based on their potential to reduce GHG emissions at the chain level.

Farmers need an incentive to implement strategies to reduce emissions. So far, agriculture does not participate in carbon trading, but this might change in the future. Other examples of (political) incentives to reduce emissions are a carbon tax or an emission quota. A policy to reduce emissions at the farm level, however, is likely to result in strategies that reduce GHG emissions on the farm, but increase GHG emissions related to the production of farm inputs. Results of this thesis showed that an increase in emissions related to farm inputs can off-set the reduction in emissions at the farm level, which might counter the intended effect of the policy. Golub et al. (2013) stress that agriculture will be affected not only by its own climate policy, but also by climate policies in forestry, energy, transport, and other sectors. Market interaction (competition for land and carbon allowances) can result in pollution swapping with other sectors. To avoid pollution swapping with other sectors, reducing GHG emissions requires an economy wide approach (Golub et al., 2012; Jensen et al., 2012).

5 Conclusions

Differences in methods and data to calculate GHG emissions from feed production can significantly affect emissions per kg feed. To harmonize methods, focus should be on calculating leaching of nitrate and on calculating GHG emissions from changes in crop management and land use.

Evaluation of strategies to reduce GHG emissions from dairy production requires an integrated approach that combines a whole-farm optimization model with a mechanistic model of enteric methane and life cycle assessment. In addition, evaluation of strategies requires a sensitivity analysis on GHG emissions from feed production, including land use change.

Each feeding strategy evaluated in this thesis reduced GHG emissions along the milk-production chain. Supplementing diets with nitrate resulted in the greatest reduction. Reducing the maturity stage of grass and grass silage was most cost-effective, and, therefore, offers most potential to be implemented on commercial farms.

Comparing the reference situations across chapters revealed that supplementing diets with urea, reducing or eliminating safety margins for rumen degradable protein and true protein digested in the small intestine, and increasing animal and plant productivity, are economically viable strategies to reduce GHG emissions from dairy production. Reducing the amount of concentrates in the diets, furthermore, provides potential to reduce GHG emissions.

The impact of increasing milk yield and longevity on GHG emissions per kg milk depends on the method that is used to handle the relation between milk and meat production.

The GHG value of milk yield is greater than that of longevity, especially when farm management is optimized based on maximizing labor income. When reducing GHG emissions becomes more important than maximizing labor income, the importance of longevity relative to milk yield increases.

When farm management is optimized based on minimizing GHG emissions, the reduction in GHG emissions, resulting from an increase in milk yield and longevity, is at least twice as great than when farm management is optimized based on maximizing labor income.

When the objective is to maximize labor income, the GHG value of milk yield relative to that of longevity decreases with increase in farm efficiency. When the objective is to minimize GHG emissions per kg milk, milk yield and longevity are equally important, independent on the level of farm efficiency.

From this thesis, we see that a combination of strategies is required to substantially reduce greenhouse gas emissions from dairy production.

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Appendices

Appendix 2.a Technical in- and output data for cultivation, drying and processing of feed crops.

Crop		Wheat (FR)[a]	Wheat (NL)[b]	Wheat (SE)[c]	Maize grain (FR)[a]	Soy beans (BR)[d]	Palm fruit (MY)[e]	Rape seed (FR)[a]	Rape seed (SE)[c]	Sugar beets (NL)[b]
Cultivation										
Yield	kg product/ha	7080	8075	7470	10483	2468[g]	21210	3300	3920	66500
Dry matter content	kg DM/kg product	0.85	0.84	0.83	0.73	1		0.92	0.88	
Synthetic fertilizer	kg N/ha	165	143	135	150	6	104	165	165	66
	kg P_2O_5-P/ha	12	3	6	25	35	70	22	7	10
	kg K_2O-K/ha	20	19	8	52	74	204	41	16	89
	kg $CaCO_3$-CaO/ha	167	0	0	167	499	0	167	0	0
Organic manure	kg N/ha	10	89	28	50	2	0	16	28	116
Insecticides	kg act sub/ha	0.0023	4	0.012	0.117	1[f]	0.31	0.02	0.07	0
Herbicides	kg act sub/ha	1.7	1.1	0.8	0.9	4.2[f]	2.4	1.0	0.7	5.1
Fungicides	kg act sub/ha	0.2	5.1	0.3	0	0.55[f]	0.013	0.08	0.042	0.4
Irrigation	m^3/ha	0	0	0	403	0	2100	0	0	0
Deposition	kg N/ha	0	29	9	0	0	18	0	9	29
Fixation	kg N/ha	0	1	0	0	130	0	0	0	1
Diesel	liter/ha	99	156	83	98	91[a]	58	109	96	101
Lubricating oil	kg/ha	0	0	7	0	0	0	0	8	0
Machinery, tillage	kg/ha	6	4.6	4.6	7	0	0.4	6	4	11
Machinery, general	kg/ha	2.7	11.7	11.7	5.7	18.2	2.1	4.1	15.3	6
Machinery, tractor	kg/ha	3.5	8	8	3.5	0	4.4	4.1	7.8	13.8
Mach., harvester	kg/ha	6.3	6.3	6.3	6.3	0	0	6.3	6.3	0
Transport, raw materials	km truck	100	100	100	100	320	100	100	100	100
	km train	600	600	600	600	0	600	600	600	600
Drying and processing										
Transport, feed crops	km tractor	0	0	5	0	0	0	0	5	0
	km truck	100	150	150	100	100	100	100	250	150
	km train	500	0	0	500	600	0	500	0	0
Drying										
Light fuel oil	MJ/ton dried product	NAP	179	181	493	183	NAP	NAP	200	7547[h]
Electricity	kWh/ton dried product	NAP	36	36	8	3	NAP	NAP	40	126[h]
Processing		NAP	NAP	NAP	NAP	Eco-invent[i]	Eco-invent[i]	Eco-invent[i]	Eco-invent[i]	Eco-invent[i]

References: [a] Nguyen *et al.*, 2012; [b] LMM, 2008; KWIN-AVG, 2009; [c] Flysjö *et al.*, 2008, updated by Wallman et al., 2012, results apply to South Sweden; [d] Prudencio da Silva et al., 2010; [e] Schmidt, 2007; [f] Meyer and Cederberg, 2010; [g] kg dry matter/ha; [h] Energy used for drying beet pulp; [i] According to inventory data for processing soybeans, palm fruit, rape seed, and sugar beets from Eco-invent (2007).

NAP = not applicable

Appendix 2.b Yield and allocation factors of crop products per crop.

Crop	Crop products	Yield kg/ha	Allocation factor
maize (FR)	maize grain	10483	1
wheat (FR)	wheat grain	7080	1
wheat (NL)	wheat grain	8075	0.87[a]
	wheat straw	3766	0.13
wheat (SE)	wheat grain	7470	0.91[b]
	wheat straw	4760	0.09

Crop	Crop products	Yield kg/t crop[c]	Allocation factor[c]
palm fruit (MY)	palm oil	216	0.81
	palm kernel oil	27	0.17
	palm kernel expeller	32	0.01
rapeseeds (FR)	rapeseed oil	396	0.75
	rapeseed meal	604	0.25
rapeseeds (SE)	rapeseed oil	396	0.75
	rapeseed meal	604	0.25
soybeans (BR)	soybean oil	182	0.41
	soybean meal	758	0.59
sugar beets (NL)	sugar	157	0.91
	molasses	36	0.05
	beet pulp	204	0.04

References: [a] KWIN-AVG, 2009; [b] Hushållningssällskapets, 2009; [c] Eco-invent, 2007.

Appendix 2.c Fertilizer specific emission factors (EFs) for NH_3 emission from synthetic fertilizer-N, and manure-N per feed crop (kg NH_3-N/kg N).

	Synthetic fert. N[a]	Manure N[b]
wheat (FR)	0.063[c]	0.076
wheat (NL)	0.02	0.082
wheat (SE)	0.02	0.2
maize (FR)	0.02	0.076
soybean meal (BR)	0.15	0.2
palm kernel expeller (MY)	0.10[c]	0.2
rape seed meal (FR)	0.066[c]	0.076
rape seed meal (SE)	0.02	0.2
beet pulp (NL)	0.02	0.082

[a] EFs based on Eco-invent (2007).

[b] EFs for feed crops from France were based on Payraudeau et al. (2007), and for the Netherlands on Huijsmans et al. (2011). For other feed crops, no specific EFs were available and IPCC Tier 1 was used.

[c] Weighted average of different types of synthetic fertilizer.

Appendix 2.d Emissions and C-uptake resulting from land use changes, and differences between studies in type of emissions accounted for (V), or not (-).

Type of emissions and C-uptake	Study 1	Study 2	Study 3	Study 4	Study 5
CO_2 from above ground biomass	V	V	V	V	V[a]
N_2O and CH_4 from burning biomass	-	V	V	-	V
CO_2 from below ground biomass	V	V	V	-	-
CO_2 from dead organic matter	V	V	-	V	-
CO_2 from changes in soil-C	V	V	V	V	V
N_2O from changes in soil-N	-	-	-	V	-
C-uptake by afforestation	V	-	V	-	-

Studies: 1 = Audsley et al. 2010, based on results from Barker et al., 2007; 2 = Leip et al., 2010; 3 = Cederberg et al., 2011; 4 = PAS 2050; 5 = Jungbluth et al., 2007.

[a] Only CO_2 from burning; 20% of total above ground biomass is assumed to be burned.

Appendix 2.e Carbon stocks in different land use categories in t C/ha (IPCC, 2006).

Land use category	Above ground-C	Below ground-C	Soil-C	Dead organic matter-C
Tropical forest (BR, MY)[a,b]	141	52	60	4
Scrubland (BR)[a]	38	15	60	0
Natural grassland (BR)[a]	3	5	60	0
Annual cropland[a,b]	0	0	35[c]	0
Perennial cropland[a]	10	NAV	60[c]	0
Secondary forest[b]	28	*5*	60[d]	0

[a] Land use categories used for annual balance method.

[b] Land use categories used for net committed emissions method.

[c] Calculated based on relative soil stock change factors (IPCC, 2006).

[d] No change in soil-C stock assumed.

NAV = not available

Appendix 3.a Emission factors for CH_4 and N_2O emissions, NO_3^- leaching, and NH_3 + NO_x volatilization from manure and managed soils at farm level.

Manure in stable/storage		Unit	Reference
CH_4	0.746	kg/ton manure	De Mol and Hilhorst, 2003.
NH_3-N	0.1	kg/kg TAN[1]	De Vries et al., 2011.
NO_x-N	0.0015	kg/kg TAN[1]	Oenema et al., 2001.
N_2O-N direct	0.0015	kg/kg TAN[1]	De Vries et al., 2011.
Managed soils			
> Grassland			
synthetic fertilizer (CAN)			
NH_3-N	0.025	kg/kg N	De Vries et al., 2011; Huijsmans and Hol, 2010.
NO-N	0.0055	kg/kg N	Stehfest and Bouwman, 2006.
N_2O-N direct	0.01	kg/kg N	Velthof and Mosquera, 2011.
slurry spreading			
NH_3-N	0.19	kg/kg TAN[1]	De Vries et al., 2011; Huijsmans and Hol, 2010.
NO_x-N	0.21	kg/kg N_2O-N direct	Eco-invent, 2007.
N_2O-N direct	0.003	kg/kg N	Velthof and Mosquera, 2011.
crop residues			
N-crop residues	48.58	kg N/ha/year	IPCC, 2006 (grassland renewal every 5 years; Aarts et al., 2002).
NO_x-N	0.21	kg/kg N2O-N direct	Eco-invent, 2007.
N_2O-N direct	0.01	kg/kg N	IPCC, 2006.
manure from grazing			
NH_3-N	0.12	kg/kg mineral N	Goossensen and Van den Ham, 1992.
NO_x-N	0.21	kg/kg N_2O-N direct	Eco-invent, 2007.
N_2O-N direct	0.025	kg/kg N minus NH_3-N	Schils et al., 2006b.
other N inputs			
N-deposition	49	kg N/ha	PBL, 2008.
N-fixation	20	kg N/ha	Schröder et al., 2004; Fraters et al., 2007.
leaching			
NO_3^--N	0.28	kg/kg N surpluss[2]	Schröder, 2005.

Appendix 3.a Continued. Emission factors for CH_4 and N_2O emissions, NO_3^- leaching, and NH_3 + NO_x volatilization from manure and managed soils at farm level.

Managed soils		Unit	Reference
> Arable land			
synthetic fertilizer (CAN)			
NH_3-N	0.025	kg/kg N	De Vries *et al*., 2011; Huijsmans and Hol, 2010.
NO-N	0.0055	kg/kg N	Stehfest and Bouwman, 2006.
N_2O-N direct	0.01	kg/kg N	Velthof and Mosquera, 2011.
slurry spreading			
NH_3-N	0.02	kg/kg TAN[1]	De Vries *et al*., 2011; Huijsmans and Hol, 2010.
NO_x-N	0.21	kg/kg N_2O-N direct	Eco-invent, 2007.
N_2O-N direct	0.013	kg/kg N	Velthof and Mosquera, 2011.
crop residues			
N-crop residues	28.39	kg N/ha/year	IPCC, 2006 .
NO_x-N	0.21	kg/kg N_2O-N direct	Eco-invent, 2007.
N_2O-N direct	0.01	kg/kg N	IPCC, 2006.
other N inputs			
N-deposition	49	kg N/ha	PBL, 2008.
leaching			
NO_3^--N	0.75	kg/kg N surpluss[2]	Schröder, 2005.
All			
N_2O-N indirect	0.01	kg/kg NH_3-N + NO_x-N	IPCC, 2006.
	0.0075	kg/kg NO_3^--N	IPCC, 2006.

[1] Total Ammoniacal Nitrogen
[2] N surplus is calculated as N-inputs minus N-outputs, i.e. (N synt. fert + N manure + N deposition + N fixation) - (N harvested crop products + N emissions)

Appendix 3.b Greenhouse gas emissions from production of farm inputs[1] and emissions from off-farm land use change (LUC) per input using two methods[2] (all in kg CO_2e/unit).

Input		Unit	Production	Off-farm LUC Method 1	Off-farm LUC Method 2[3]
agricultural operations[4]	application of plant protection products, by field sprayer	ha	10.94	-	0.00
	chopping, maize	ha	320.06	-	0.01
	fertilizing, by broadcaster	ha	25.24	-	0.00
	haying, by rotary tedder	ha	10.76	-	0.00
	hoeing	ha	20.43	-	0.00
	mowing, by rotary mower	ha	23.26	-	0.00
	sowing	ha	22.65	-	0.00
	swath, by rotary windrower	ha	16.14	-	0.00
	tillage, cultivating, chiseling	ha	71.11	-	0.00
	tillage, harrowing, by rotary harrow	ha	62.25	-	0.00
	tillage, harrowing, by spring tine harrow	ha	24.62	-	0.00
	tillage, ploughing	ha	118.20	-	0.00
	tillage, rolling	ha	23.43	-	0.00
	fodder loading, by self-loading trailer	m3	0.62	-	0.07
	slurry spreading, by vacuum tanker	m3	1.20	-	0.15
synthetic fertilizer	calcium ammonium nitrate (CAN), as N	kg N	8.65	-	0.11
	potassium chloride, as K2O	kg K_2O	0.50	-	0.05
	triple superphosphate, as P2O5	kg P_2O_5	2.01	-	0.09
	urea, as N	kg N	3.30	-	0.06
	quicklime, milled, loose at plant	kg	0.98	-	0.00
pesticides	insecticides	kg	16.57	-	0.37
	herbicides	kg	10.12	-	0.26
	fungicides	kg	10.51	-	0.23
	unspecified	kg	9.99	-	0.25

Appendix 3.b Continued. Greenhouse gas emissions from production of farm inputs[1] and emissions from off-farm land use change (LUC) per input using two methods[2] (all in kg CO_2e/unit).

Input		Unit	Production	Off-farm LUC Method 1	Off-farm LUC Method 2[3]
diesel	at regional storage	kg	0.51	-	0.00
	combustion	kg	3.15	-	-
	supply and combustion	kg	3.66	-	0.00
gasoline	light fuel oil, at regional storage	MJ	0.01	-	0.00
	combustion	MJ	0.07	-	-
	supply and combustion	MJ	0.09	-	0.00
electricity	low voltage (househ. & agri.) at grid	MJ	0.20	-	0.00
natural gas	low pressure, at consumer	MJ	0.02	-	0.00
	combustion	MJ	0.06	-	0.00
	supply and combustion	MJ	0.07	-	-
tap water		m^3	0.10	-	0.00
concentrates	standard protein	ton DM	719	116.88	105.09
	medium protein	ton DM	741	113.74	110.40
	high protein	ton DM	819	144.50	148.22
	rearing	ton DM	719	116.88	105.02
	heifers	ton DM	719	116.88	105.02
roughage	maize silage	ton DM	212	-	100.52
milk replacer		ton DM	1904.03	-	4.13
litter	sawdust	ton DM	66.21	-	1.82

[1] Emission factors are based on Eco-invent (2007), except for concentrates (Vellinga et al., 2012), purchased maize silage (this study), and milk replacer and litter(Thomassen et al., 2008).

[2] Emissions from off farm LUC such as deforestation. Method 1 includes direct LUC only (Prudencio da Silva et al., 2010, Jungbluth et al., 2007, Van Middelaar et al., 2013a). Method 2 includes direct and indirect LUC (Audsley et al., 2009).

[3] LUC emissions are computed by multiplying the amount of hectares used for producing the inputs (Eco-invent, 2007; Vellinga et al., 2012) with the emission factor from Audsley et al. (2009), i.e.1.43 t CO_2e/ha. As for the production of machinery, fertilizer, and pesticides land is required as well, this method assigns LUC emissions to the production of these inputs. Land use per concentrate type: standard protein 0.0735 ha/t DM; medium protein 0.0772 ha/t DM; high protein 0.1036 ha/t DM.

[4] Emissions include emissions from production of diesel and machinery (approximately 50%), and from combustion of diesel (about 50%). The latter is included in on-farm emissions.

Appendix 3.c Inventory data and total GHG emissions from the production of purchased maize silage[1]

Cultivation and transport	Unit		Reference[2]
gross yield	kg DM/ha	13250	1
harvesting and feeding losses	%	8	1
synthetic fertilizer	kg N/ha	33	2
	kg P_2O_5/ha	16	3
	kg K_2O/ha	75	3
organic manure	kg N/ha	182	2
Pesticides - insecticides	kg act sub/ha	0	3
- herbicides	kg act sub/ha	2	3
- fungicides	kg act sub/ha	0	3
N deposition	kg N/ha	31	4
tillage, by spring tine harrow	# activities	2	5
tillage, ploughing	# activities	1	5
sowing	# activities	1	5
appl. of plant protection prod.	# activities	1	5
chopping maize	# activities	1	5
haying	# activities	1	5
fodder loading	m^3/ha	190	5
fertilizing (synth. fert)	no. activities	3	5
transport from production site to dairy farm	tkm	203	6
Total GHG emissions	kg CO_2e/t DM	212.3	

[1] Inventory data for the production of home grown maize silage (on-farm roughage production) are similar to the data presented in this appendix, except for the use of fertilizer N. In the dairy farm LP model, it was assumed that maize land was fertilized with 150 kg N/ha. The type of N-fertilizer that was used (manure, or synthetic) resulted from the LP model.

[2] References: 1.KWIN-V, 2008; 2.LMM, 2008; 3.KWIN-AGV, 2009; 4. PBL, 2008; 5. Eco-invent, 2007; 6. Estimated average from production site to farm: 20 km.

Appendix 5.a Nutritional values of feed ingredients[1]

Dietary options	NE_L[2] (MJ/kg DM)	DVE[3] (g/kg DM)	OEB[4] (g/kg DM)	N (g/kg DM)	P (g/kg DM)	Fill value[5] (kg/kg)
Concentrates						
- standard protein	7.2	100	5.6	24.1	4.5	0.29-0.72
- medium protein	7.2	133	27.8	32.2	5.0	0.29-0.72
- high protein	7.2	200	83.3	48.3	8.0	0.29-0.72
Fresh grass						
- 125 kg N	6.6	94	9.3	28.0	4.1	0.93
- 175 kg N	6.7	96	16.1	29.4	4.1	0.93
- 225 kg N	6.7	98	23.5	30.9	4.1	0.93
- 275 kg N	6.8	99	31.2	32.4	4.1	0.93
Grass silage						
- 125 kg N	5.9	70	22.2	25.6	4.1	1.08
- 175 kg N	5.9	71	30.6	27.4	4.1	1.08
- 225 kg N	6.0	73	39.0	29.0	4.1	1.08
- 275 kg N	6.0	74	47.3	30.6	4.1	1.08
Maize silage	6.6	58	-36.0	13.4	1.9	1.02

[1] All feed ingredients are available in summer and winter, except for fresh grass (only in summer).
[2] Net energy for lactation.
[3] True protein digested in the small intestine according to Dutch standards (Tamminga et al., 1994).
[4] Rumen degradable protein balance according to Dutch standards (Tamminga et al., 1994).
[5] Fill value per kg feed expressed in kg of a standard reference feed (see Jarrige, 1988). The fill value of concentrates increases with an increase in concentrate intake.

Appendix 5.b Emissions factors for enteric fermentation and production per feed product.

Feed product	Enteric fermentation[1] gram CH_4/kg DM	Production[2] gram CO_2e/kg DM Excl. LUC[3]	Incl. LUC
Grazed grass			
- 125 kg N/ha	23.74	128 – 284	313 – 469
- 175 kg N/ha	22.74	142 – 337	309 – 504
- 225 kg N/ha	21.74	156 – 386	311 – 541
- 275 kg N/ha	20.74	171 – 435	316 – 580
Grass silage			
- 125 kg N/ha	22.74	153 – 301	329 – 477
- 175 kg N/ha	21.74	164 – 349	322 – 507
- 225 kg N/ha	20.74	175 – 393	322 – 540
- 275 kg N/ha	19.74	187 – 438	325 – 575
Maize silage home grown	17.74	157 – 229	251 – 323
Maize silage purchased	17.74	176	267
Concentrates			
- standard protein	20.09	724	836
- medium protein	19.55	741	862
- high protein	19.93	774	934

[1] Emission factors are based on Vellinga et al (2013), using a mechanistic model originating from Dijkstra et al. (1992) and updated by Mills et al. (2001) and Bannink et al. (2006).

[2] Emissions from production of home grown grass and maize silage depend on type of fertilizer (organic or synthetic), which is an outcome of the LP model. Emissions per kg DM can vary between the lowest (100% organic fertilizer) and the highest (100% synthetic fertilizer) figure. Emissions factors are presented excluding and including emissions from land use change. Including LUC involves an additional emission of 1.43 t CO_2e/ha due to deforestation worldwide (Audsley et al., 2009). Land use per ton DM maize silage is 0.061 ha, and per ton DM concentrates 0.087 ha (standard protein), 0.094 ha (medium protein), and 0.125 ha (high protein).

[3] LUC = land use change.

Appendix 5.c Live weight and amount of edible product per animal, and emission factors per kg edible product related to cows and calves culled from the dairy farm.

	Live weight[1] (kg/animal)	Edible product[2] (kg/animal)	Emission factor[3] (kg CO_2e/kg edible product)
Dairy cows	650	264	0.4
Heifers	545	221	0.4
Young stock > 12 months	320	130	0.4
Calves (white veal)	225	101	10.5

[1] Based on Remmelink et al. (2012).

[2] The ratio between edible product / live weight were estimated based on the ratio for beef (0.43) (De Vries and De Boer, 2010) and the ratio between live weight and carcass weight for dairy cattle according to Sebek and Temme (2009) and KWIN (2013).

[3] Emissions from transport and slaughtering of dairy cows, heifers, and young stock > 12 months were based on *The Finnish Environment* 539 (2002). Emissions from production of white veal (emissions from feeding, housing, transport and slaughter) are based on H. Mollenhorst (Wageningen UR, Wageningen, The Netherlands, personal communication).

Summary

The dairy sector contributes to climate change, mainly through the emission of greenhouse gases (GHGs) carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O). Emissions of CO_2 relate to fossil fuel combustion and land use change, emissions of CH_4 to enteric fermentation and manure management, and emissions of N_2O to manure management and application of nitrogen fertilizer for cultivation of feed crops. Main sources of GHG emissions from dairy production are enteric fermentation, feed production, and manure management. Important areas of interest to reduce GHG emissions per kg milk include feeding strategies to reduce emissions from enteric fermentation and feed production, and breeding strategies to improve animal productivity.

Most studies that evaluate the potential of feeding and breeding strategies to reduce GHG emissions from dairy production do not account for emissions other than enteric CH_4, do not account for changes in farm management to adapt the farm optimally to the particular strategy, or do not account for consequential effects in other parts of the milk-production chain. The two objectives of this thesis were to develop an integrated method to evaluate strategies to reduce GHG emissions from dairy production at the chain level, and to evaluate feeding and breeding strategies using this integrated method.

Effective evaluation of strategies requires accurate methods and data to calculate emissions from each process along the chain. To improve comparison and interpretation of GHG calculations of feed products, **Chapter 2** explored consequences of differences in methods and data to calculate GHG emissions per kg feed. Methods and data to calculate emissions from cultivation and processing, and from changes in crop management and land use were analyzed for six concentrate ingredients (i.e. maize, wheat, palm kernel expeller, rapeseed meal, soybean meal, and beet pulp). Results showed that methods to calculate nitrate leaching, which is required to estimate indirect N_2O emissions, and methods to calculate emissions from changes in land use vary, and that differences significantly affect results. In addition, GHG calculations are not robust to assumptions on crop yield per hectare and use of synthetic fertilizer (i.e. nitrogen).

In **Chapter 3** the relevance of integrated modeling was demonstrated by evaluating the impact of increasing maize silage at the expense of grass and grass silage in a dairy cow's diet at three levels: animal, farm, and chain levels. A whole-farm optimization model was combined with a life cycle approach and a mechanistic model to predict enteric CH_4 production. The impact of the strategy was evaluated for a typical Dutch dairy farm on sandy soil. First, the whole-farm model was used to define a reference situation without a predefined feeding strategy. Optimization of farm management was based on maximizing labor income. Subsequently, maize silage was increased by 1 kg dry matter/cow per day at the expense of grass (summer), or grass silage (winter). Then, the model was optimized again to determine the new economically optimal farm situation. Emissions of GHGs in the reference situation and after implementing the strategy were calculated and compared at animal, farm, and chain levels. Emissions were allocated to milk and meat based on economic allocation. Results showed that the level of analysis strongly influences results and conclusions, and confirmed the importance of integrated modeling. At animal level, the strategy resulted in an immediate reduction of GHG emissions (12 kg CO_2 equivalents (CO_2e)/ton fat-and-protein-corrected milk (FPCM)). Analysis at farm and chain levels revealed that the strategy is not feasible on farms that cannot further reduce their grassland area because of compliance with the EU derogation regulation. For more intensive farms that can reduce their grassland area, it takes 61 years at chain level, before annual emission reduction (14 kg CO_2e/ton FPCM) has paid off non-recurrent emissions from changing grassland into maize land (860 kg CO_2e/ton FPCM).

After demonstrating the relevance of an integrated method, the method was used to evaluate the impact of several feeding and breeding strategies on GHG emissions at chain level and on labor income at farm level. In **Chapter 4** the cost-effectiveness of three feeding strategies was evaluated. Strategies included were: supplementing diets with an extruded linseed product (56% linseed; 1 kg/cow per day in summer and 2 kg/cow per day in winter), supplementing diets with a nitrate source (75% nitrate; 1% of DM intake), and reducing the maturity stage of grass and grass silage (grazing at 1400 instead of 1700 kg DM/ha and harvesting at 3000 instead of 3500 kg DM/ha). Total GHG emissions in the reference situation were 840 kg CO_2e per/t FPCM. Linseed supplementation reduced emissions by 9 kg CO_2e/ton FPCM, nitrate supplementation by 32 kg CO_2e/ton FPCM, and reducing grass maturity by 11 kg CO_2e/ton FPCM. Of all three strategies, reducing grass maturity was most cost-effective (i.e. GHG reduction costs were €57/ton CO_2e compared to €241/ton CO_2e for nitrate and €2594/ton CO_2e for linseed).

To evaluate the potential of breeding strategies to reduce GHG emissions, **Chapter 5** explored methods to calculate GHG values of genetic traits. The GHG value of a genetic trait represents the impact of one genetic standard deviation improvement on GHG emissions at chain level while keeping the other traits constant. GHG values provide insight into the relative importance of individual traits to reduce GHG emissions along the chain and can be used to include environmental performance of traits in breeding programs. GHG values of milk yield and longevity were calculated based on two different optimization methods. The first method optimized farm management before and after a change in genetic trait by maximizing labor

income; the impact on GHG emissions was considered as a side effect. The second method optimized farm management after a change in genetic trait by minimizing GHG emissions per kg milk, while maintaining labor income and milk production at least at the level before the change in trait. In the reference situation, milk yield per cow was assumed to be 8758 kg/year and longevity 2150 days. The genetic standard deviation (σ_g) for milk yield is 687 kg/cow per year, and for longevity 270 days. Results showed that GHG values of milk yield and longevity were respectively 247 and 210 kg CO_2e/σ_g per cow per year when maximizing labor income, and 453 and 441 kg CO_2/σ_g per cow per year when minimizing GHG emissions. So, GHG values of both traits were about twice as great when minimizing GHG emissions than when maximizing labor income. In addition, the GHG value of milk yield was greater than that of longevity, especially when maximizing labor income. Results indicate that the importance of longevity relative to milk yield increases as importance of reducing GHG emissions increases.

The GHG values calculated in Chapter 5 concerned a typical dairy farm in 2020, with a high efficiency concerning feed utilization and feed production. To evaluate the robustness of GHG values to assumptions on feed-related farm characteristics, **Chapter 6** calculated GHG values of milk yield and longevity for a less efficient farm and compared those with the values from Chapter 5. The less efficient farm did not apply precision feeding and had lower grass and maize yields per ha than the efficient farm in Chapter 5. On the less efficient farm, GHG values of milk yield and longevity were respectively 279 and 143 kg CO_2e/σ_g per cow per year when maximizing labor income, and 538 and 563 kg CO_2/σ_g per cow per year when minimizing GHG emissions. Thus, when maximizing labor income, the importance of milk yield relative to longevity increased with decrease in farm efficiency. When minimizing GHG emissions, GHG values of both traits increased with decrease in farm efficiency, but the ratio between values hardly changed. When minimizing GHG emissions, both traits were equally important on each level of efficiency.

In **Chapter 7** the relevance and methodological challenges of integrated modeling were discussed. The discussion revealed that effective evaluation of strategies to reduce GHG emissions from dairy production requires an integrated method that combines a whole-farm optimization model with a life cycle approach and a mechanistic model to predict enteric CH_4 production. In addition, the importance of including a sensitivity analysis to account for variability and uncertainty in GHG calculations of feed products was addressed, as well as the impact of different methods for handling the relation between milk and meat production. Comparison of results across chapters revealed several other strategies to reduce GHG emissions from dairy production, including supplementing diets with urea, reducing safety margins for rumen degradable protein and protein digested in the small intestine, increasing animal and plant productivity, and reducing the amount of concentrates in the diet. To substantially reduce GHG emissions from dairy production, a combination of different strategies is required.

Chapter 7 ends with some practical implications for reducing GHG emissions on commercial farms. Integrated modeling can be used to identify most promising strategies to reduce GHG emissions while maintaining a decent income at farm level. In case of breeding, GHG values of genetic traits, other than milk yield and longevity, need to be estimated in order to define and evaluate impacts of multi-trait selection strategies. For successful implementation of strategies on commercial farms, variation between farms has to be acknowledged. To identify farm specific strategies, a benchmark or decision support tool is required. A policy to stimulate farmers to reduce GHG emissions (e.g. a carbon tax or emission quota) requires an economy wide approach to prevent pollution swapping within and between sectors.

Samenvatting

De melkveehouderij draagt bij aan de uitstoot van broeikasgassen, waarvan koolstofdioxide (CO_2), methaan (CH_4), en lachgas (N_2O) de belangrijkste zijn. De uitstoot van CO_2 is voornamelijk gerelateerd aan de verbranding van fossiele energie en ontbossing; de uitstoot van CH_4 aan pensfermentatie (enterische CH_4) en mestmanagement; en de uitstoot van N_2O aan mestmanagement en het gebruik van stikstof uit kunstmest en organische mest tijdens de teelt van voedergewassen. De uitstoot van broeikasgassen draagt bij aan klimaatverandering en aan de gevolgen die daarmee gepaard gaan, zoals extreme weersomstandigheden en verlies aan biodiversiteit. Om de uitstoot van broeikasgassen in de melkveehouderij te verlagen worden verschillende maatregelen voorgesteld, waaronder voermaatregelen en fokkerijmaatregelen.

De meeste studies die het effect van voer- en fokkerijmaatregelen analyseren richten zich enkel op de uitstoot van enterische CH_4, houden geen rekening met veranderingen in het management op het melkveebedrijf ten gevolge van de maatregel, of houden geen rekening met de consequenties die de maatregel heeft op de uitstoot van broeikasgassen tijdens andere processen in de keten. Het doel van deze studie is om een integraal model te ontwikkelen om het effect van verschillende voer- en fokkerijmaatregelen op ketenniveau te bepalen.

Het analyseren van maatregelen op ketenniveau vereist methoden en data om de uitstoot van broeikasgassen tijdens alle processen in de keten te kunnen bepalen. Methoden en data voor het bepalen van de broeikasgasuitstoot tijdens de productie van voer bevatten veel variatie en onzekerheid. In **hoofdstuk 2** is onderzocht wat de consequenties zijn van deze variatie en onzekerheid voor het schatten van de uitstoot van broeikasgassen tijdens de productie van zes verschillende voeringrediënten: maïs, tarwe, palmpitschroot, raapzaadschroot, sojaschroot en bietenpulp. Resultaten tonen aan dat methoden om nitraatuitspoeling te schatten (dat is nodig om de indirecte uitstoot van N_2O te bepalen), en methoden om de uitstoot ten gevolge van landgebruiksveranderingen te schatten veel variatie bevatten en resultaten sterk beïnvloeden. Daarnaast blijken schattingen voor de uitstoot van broeikasgassen niet robuust voor aannames ten aanzien van de gewasopbrengst en het gebruik van kunstmest (stikstof) tijdens de teelt.

In **hoofdstuk 3** wordt het belang van een integrale methode voor het evalueren van maatregelen gedemonstreerd. Drie modellen worden gecombineerd: een mechanistisch pensmodel, een optimalisatiemodel van een melkveebedrijf (lineaire programmering), en levenscyclusanalyse. Met het integrale model wordt het effect van een voermaatregel om enterische CH_4 te verlagen doorgerekend op zowel dier-, bedrijfs-, als ketenniveau. De maatregel betreft het uitwisselen van gras- voor maïssilage in het rantsoen van koeien, en effecten worden bepaald voor een gemiddeld Nederlands melkveebedrijf op zandgrond. Met behulp van het optimalisatiemodel wordt de referentiesituatie vastgesteld, waarbij maximalisatie van het inkomen op het melkveebedrijf als doelstelling gebruikt wordt. Vervolgens wordt de hoeveelheid maïssilage in het rantsoen verhoogd met 1 kg droge stof (DS) per koe/dag, ten koste van vers gras (zomer) of grassilage (winter). Opnieuw wordt het model geoptimaliseerd om te bepalen hoe de situatie na invoering van de maatregel eruitziet. Broeikasgasuitstoot per kg meetmelk wordt bepaald voor de referentiesituatie en na invoering van de maatregel, en gealloceerd naar melk en vlees op basis van economische allocatie. Op dierniveau leidt de maatregel tot een afname van de broeikasgasuitstoot (12 kg CO_2 equivalenten (CO_2e)/ton meetmelk). Analyse op bedrijfsniveau toont aan dat de maatregel enkel geschikt is voor bedrijven die hun areaal grasland kunnen verlagen, wat meestal niet het geval is in verband met de Europese derogatieregeling die alleen geldt voor bedrijven met minstens 70% grasland. Op intensieve bedrijven die hun areaal grasland wel kunnen verlagen duurt het 61 jaar voordat de jaarlijkse afname in uitstoot op ketenniveau (14 kg CO_2e/ton meetmelk per jaar) de eenmalige uitstoot ten gevolgen van het omploegen van grasland voor maïsland (860 kg CO_2e/ton meetmelk) overstijgt.

In **hoofdstuk 4** wordt het integrale model vervolgens gebruikt om het effect van verschillende voermaatregelen te analyseren. Door het effect op broeikasgasuitstoot te combineren met het effect op inkomen wordt voor de volgende maatregelen de kosteneffectiviteit bepaald: het voeren van een geëxtrudeerd lijnzaadproduct (56% lijnzaad, 1 kg per koe/dag in de zomer en 2 kg per koe/dag in de winter), het voeren van een nitraatproduct (75% nitraat; 1% van de totale DS opname), en het beweiden en maaien van jonger gras (beweiden bij snedeopbrengst van 1400 ipv 1700 kg DS/ha, en maaien bij 3000 ipv 3500 kg DS/ha). In de referentiesituatie komt de totale broeikasgasuitstoot uit op 840 kg CO_2e/ton meetmelk. Toevoegen van lijnzaad verlaagt deze emissie met 9 kg CO_2e/ton meetmelk, toevoegen van nitraat met 32 kg CO_2e/ton meetmelk, en het beweiden en maaien van jonger gras met 11 kg CO_2e/ton meetmelk. Van de drie strategieën blijkt deze laatste strategie het meest kosteneffectief: de kosten voor het verlagen van de uitstoot door het beweiden en maaien van jonger gras liggen op €57/ton CO_2e, tegenover €241/ton CO_2e voor het voeren van nitraat, en €2594/ton CO_2e voor het voeren van lijnzaad.

Om het effect van fokkerijmaatregelen te analyseren worden in **hoofdstuk 5** twee methoden getest om broeikasgaswaarden van genetische kenmerken te bepalen. De broeikasgaswaarde van een kenmerk geeft het effect weer van een verandering in dat kenmerk ter grootte van één genetische standaarddeviatie op de uitstoot van broeikasgassen in de keten. Alle andere kenmerken blijven tijdens deze evaluatie gelijk. Broeikasgaswaarden geven inzicht in het relatieve belang van individuele kenmerken om broeikasgassen te verlagen en kunnen worden gebruikt door fokkerijorganisaties die milieuprestaties in het fokprogramma willen meenemen. De broeikasgaswaarden van de kenmerken melkproductie en levensduur zijn bepaald op basis de volgende twee methoden. Methode 1 optimaliseert het melkveebedrijf vóór en na een genetische verandering op basis van maximalisatie van het inkomen; de verandering in broeikasgassen wordt beschouwd als een neveneffect. Methode 2 optimaliseert het melkveebedrijf na een genetische verandering op basis van minimalisatie van broeikasgassen per kg melk, terwijl het inkomen en de totale melkproductie op het bedrijf minstens gelijk blijven aan het niveau vóór de verandering. Melkproductie per koe is in de referentiesituatie 8758 kg/jaar en levensduur 2150 dagen. De genetische standaarddeviatie (σ_g) van melkproductie is 687 kg melk/koe per jaar, en van levensduur 270 dagen. Resultaten tonen aan dat de broeikasgaswaarden voor melkproductie en levensduur respectievelijk 247 en 210 kg CO_2e/σ_g per koe per jaar zijn in het geval van methode 1, en 453 en 441 kg CO_2e/σ_g per koe per jaar in het geval van methode 2. De broeikasgaswaarden van beide kenmerken zijn dus ongeveer twee keer zo groot wanneer de nadruk ligt op het minimaliseren van broeikasgassen dan wanneer de nadruk ligt op het maximaliseren van inkomen. Daarnaast leidt een verhoging van de melkproductie tot een grotere afname in broeikasgassen dan een verlenging van de levensduur, vooral wanneer maximalisatie van het inkomen centraal staat. Dit geeft aan dat het belang van levensduur ten opzichte van melkproductie toeneemt wanneer het verlagen van broeikasgassen belangrijker wordt.

In hoofdstuk 5 zijn de broeikasgaswaarden van melkproductie en levensduur uitgerekend voor een typisch Nederlands melkveebedrijf in 2020, met een hoge efficiëntie aangaande de productie en het gebruik van voer. Om te bepalen hoe robuust deze waarden zijn, worden in **hoofdstuk 6** dezelfde broeikasgaswaarden uitgerekend voor een minder efficiënt bedrijf en vergeleken met de resultaten uit hoofdstuk 5. Het minder efficiënte bedrijf heeft een lagere voeropbrengst per ha dan het efficiënte bedrijf, en voert boven de voedernorm voor eiwit. Op het minder efficiënte bedrijf zijn de broeikasgaswaarden van melkproductie en levensduur respectievelijk 279 en 143 kg CO_2e/σ_g per koe per jaar in het geval van methode 1, en 538 en 563 kg CO_2e/σ_g per koe per jaar in het geval van methode 2. Deze resultaten tonen aan dat wanneer maximalisatie van het inkomen centraal staat, het belang van melkproductie ten opzicht van levensduur toeneemt wanneer de efficiëntie van het bedrijf afneemt. Wanneer minimalisatie van broeikasgassen centraal staat, nemen de broeikasgaswaarden van beide kenmerken toe wanneer de efficiëntie van het bedrijf afneemt, maar de ratio tussen de waarden van de kenmerken blijft nagenoeg gelijk: beide kenmerken zijn ongeveer even belangrijk voor het verlagen van broeikasgassen ongeacht de efficiëntie van het bedrijf.

In **hoofdstuk 7** worden de relevantie en de methodische uitdagingen van een integrale methode bediscussieerd. Een correcte evaluatie van maatregelen vereist een methode waarbij een optimalisatiemodel wordt gecombineerd met een mechanistisch pensmodel en een ketenbenadering. Daarnaast is het belangrijk om met behulp van een gevoeligheidsanalyse het effect van variatie en onzekerheid in broeikasgasberekeningen van voerproductie te bepalen. Vergelijking van de resultaten over de hoofdstukken laat zien dat er verschillende andere mogelijkheden zijn om broeikasgassen te verlagen, zoals het voeren van ureum, het toepassen van precisievoeding, het verhogen van de productiviteit van zowel het melkvee als het gewas, en het verlagen van de hoeveelheid krachtvoer in het rantsoen. Uiteindelijk is er echter een combinatie van maatregelen nodig om de uitstoot van broeikasgassen in de melkveehouderij ingrijpend te verlagen.

Hoofdstuk 7 eindigt met de uiteenzetting van stappen om de uitstoot van broeikasgassen op melkveebedrijven in de praktijk te verlagen. Met behulp van een integrale methode kunnen strategieën worden geïdentificeerd die leiden tot een afname in broeikasgassen op ketenniveau en waarvan de kosten beperkt zijn. Om tot een effectief fokbeleid te komen moeten de broeikasgaswaarden van meerdere genetische kenmerken worden bepaald, waarna het effect van een compleet fokdoel kan worden doorgerekend. Daarnaast is het belangrijk om te erkennen dat ieder melkveebedrijf anders is, en dat dus ieder bedrijf om specifieke maatregelen vraagt. Om deze specifieke maatregelen te identificeren, is een tool nodig die boeren kan helpen inzicht te krijgen in hun milieuprestatie en het effect van maatregelen op hun bedrijf. Om te voorkomen dat maatregelen uiteindelijk leiden tot een toename van broeikasgassen ergens anders in de keten, of in andere sectoren, is een internationaal beleid gewenst.

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Corina van Middelaar

About the author

Corina van Middelaar was born in Amersfoort in 1984. She completed her BSc study in Animal Husbandry at HAS Den Bosch (2007), and her MSc study in Animal Sciences at Wageningen University (2009). Her first master thesis was carried out within the Entomology chair group and focused on circadian rhythm and olfactory responses of Culicoides species in the Netherlands. Her second master thesis was carried out within the Animal Production Systems chair group and focused on eco-efficiency of cheese production in the Netherlands, resulting in a scientific publication (Van Middelaar et al., 2011). After graduation she started as a PhD student within the Animal Production Systems group of Wageningen University. Her PhD research was directed at integrated modeling of feeding and breeding strategies to reduce greenhouse gas emissions from dairy production, and financed by CRV B.V., Agrifirm Group, and the Ministry of Infrastructure and the Environment. Her PhD thesis work was awarded the ADSA (American Dairy Science Association) travel award at the EAAP (European Federation for Animal Science) conference in 2011. Since completing her PhD research in March 2014 she has been working as a postdoctoral fellow with the Animal Production Systems group of Wageningen University.

Publications

Refereed scientific journals

Van Middelaar, C.E., P.B.M. Berentsen, M.A. Dolman, and I.J.M. De Boer. 2011. Eco-efficiency in the production chain of Dutch semi-hard cheese. Livest. Sci. 139:91-99

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Meul, M., C.E. Van Middelaar, I.J.M. De Boer, D. Fremaut, and G. Haesaert. 2014. Potential of life cycle assessment to support environmental decision making at commercial dairy farms. Agric. Syst. *Accepted with revisions.*

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De Boer, I.J.M., and C.E. Van Middelaar. 2012. Towards a sustainable animal production sector: potential and problems of LCA. In: Book of Abstracts of the 8th international conference on Life Cycle Assessment in the Agri-Food sector, October 1 – 4, Saint-Malo, France, p 35.

Education certificate

Completed training and supervision plan[1]

The basic package (3.0 ECTS)
- WIAS Introduction Course (2009)
- WGS Course 'Ethics and Philosophy of Animal Science' (2010)

International conferences (5.1 ECTS)
- LCA Food, Bari, Italy (2010)
- GGAA, Banff, Canada (2010)
- EAAP, Stavanger, Norway (2011)
- ADSA, Phoenix, America (2012)

Seminars and workshops (1.8 ETCS)
- WIAS Science Day, Wageningen (2010-2013)
- Symposium 'How to keep the planet alive', Ede (2010)
- Workshop 'Carbon footprints of animal feed', Schiphol (2011)

Presentations (9.0 ETCS)
- WIAS Science Day, Wageningen, poster (2010)
- LCA Food, Bari, Italy, poster (2010)
- WIAS Science Day, Wageningen, poster (2011)
- Workshop 'Carbon footprints of animal feed', Schiphol, oral (2011)
- EAAP, Stavanger, Norway, oral (ADSA/EAAP Travel Award) (2011)
- Novus International, St Louis MO, USA, oral (2012)
- ADSA, Phoenix AZ, USA, oral (2012)
- CRV Board Committee, Wageningen, oral (2012)
- WIAS Science Day, Wageningen, oral (2013)

In-depth studies (8.5 ETCS)
- MSc Course 'Nutrient Dynamics', WUR (2010)
- PhD Course 'Advanced LCA', Aalborg University, Denmark (2010)
- WIAS Course 'Tropical farming systems with livestock' (2013)
- PhD LCA discussion group, WUR (2009-2013)

Professional skills support courses (3.8 ETCS)
- PhD Competence assessment (2010)
- Techniques for Scientific Writing (2011)
- Project- and Time Management (2011)
- Supervising MSc thesis work (2012)

Research skills training (9.5 ETCS)
- Preparation own PhD research proposal (2009)
- Review scientific paper (2012)
- Supervision PhD thesis (2013)

Didactic skills training (17.5 ETCS)
- Supervision practical 'Inleiding dierwetenschappen' (2009-2010)
- Review RMC proposals (2011)
- Supervision MSc theses (2011-2012)
- Lectures and practical 'Sustainable development of animal systems' (2011-2013)
- Lectures 'Advanced Biosystems Engineering' (2011-2013)
- Preparation MSc course material 'Sustainable development of animal systems' (2013)
- Course coordinator 'Sustainable development of animal systems' (2013)

Management skills training (2.0 ETCS)
- Organization WIAS Science Day (2012)
- Design PhD course (2012)

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

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