Life cycle assessment of cattle production: exploring practices and system changes to reduce environmental impacts

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Analyse de cycle de vie de la production bovine : exploration de pratiques et de changements de système pour réduire les impacts environnementaux

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Preface

This thesis aims to address a major concern at the interface of livestock production and environmental issues. Due to the global environmental impacts of livestock, a huge research effort is necessary to provide information and, if possible, solutions in this area. The present work is a contribution to this effort. It has focused on cattle farming systems, which represent a major part of animal products and land occupation in France. It has explored several major environmental issues, and new ways to analyse them.

This thesis has been prepared in a context of close collaboration between two INRA laboratories with complementary skills. The Herbivore Research Unit is focused on ruminant physiology, nutrition and farming systems for ruminant production, aiming to improve product quality and animal welfare while reducing environmental impacts. Within this research unit, a team named "Microbial digestion and absorption" carries out research to reduce enteric methane emissions and to analyse consequences of methane mitigation strategies on emissions of other greenhouse gases and on environmental impacts. The Soil Agro- and hydroSystem (SAS) research unit mainly studies interactions among the fields of agronomy, hydrology, soil science, and environmental analysis. One of its teams (ASAE) researches ways to improve methods of assessing environmental impacts of agricultural systems, in particular life cycle assessment, and applying these assessments.

The thesis was funded by Valorex (La Messayais, 35210 Combourtillé, France), a company with two main activities: selling extruded products, mainly linseeds, and advisory services focused on the positive effects for human health of omega-3 fatty acids. As linseeds have been shown to decrease CH_4 emissions, Valorex was interested in studying the potential of omega-3 fatty acids to decrease the climate change impact. In a preliminary work, evaluation of environmental impacts from ruminants was performed for a part of a beef-production system; it is presented in the Annex of this thesis.

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Abbreviations and acronyms

ALCA	Attributional life cycle assessment
AP	Acidification potential
ASAE	Agro-environmental Analysis of Crop-Livestock Systems
CAP	Common Agricultural Policy
CC	Climate change
CED	Cumulative energy demand
CH_4	Methane
CLCA	Consequential life cycle assessment
CO_2	Carbon dioxide
DIMA	Microbial digestion and absorption research group
DM	Dry matter
EP	Eutrophication potential
EU	European Union
FAs	Fatty acids
FAO	Food and Agriculture Organization of the United Nations
GHG	Greenhouse gas
GTAP	Global Trade Analysis Project
INRA	Institut National de la Recherche Agronomique
IPCC	Intergovernmental Panel on Climate Change
LCA	Life cycle assessment
LCI	Life cycle inventory
LCIA	Life cycle inventory assessment
LEITAP	Landbouw Economisch Instituut Trade Analysis Project
LO	Land occupation
LU	Land-use
LUC	Land-use change
LULUC	Land-use and land-use change
NH_3	Ammonia
N_2O	Nitrous oxide
UMRH	Herbivores Research Unit of INRA
UMR SAS	Soil Agro and hydroSystem Research Unit of INRA
USA	United States of America

This word cloud has been generated using Wordle. The source texts used to create this cloud are the titles, abstracts and key words of Chapters 2 to 5 of the thesis. The cloud gives greater prominence to words that appear more frequently in these texts. The cloud thus reflects the key words of this thesis.

Abstract

Life cycle assessment of cattle production: exploring practices and system changes to reduce environmental impacts

This thesis addresses the environmental impacts of cattle production systems. The first objective of this thesis was to analyse and compare the environmental impacts of suckler-beef and dairy production systems using attributional life cycle assessment (ALCA). Subsequently, the effects of mitigation practices for suckler-beef production systems were assessed. The second objective addressed methodology development by exploring possible consequences due to an increase in preference for grass-based milk using consequential LCA (CLCA).

For a suckler-beef production system, enteric methane fermentation was the main contributor to the climate change impact, and grassland production contributed most to other impacts (cumulative energy demand, eutrophication, acidification and land occupation). The suckler cow-calf herd substantially contributed to the impacts of the suckler-beef system. The most effective mitigation practice for the suckler-beef production system was decreasing calving age from 3 to 2 years. The use of lipids rich in omega-3 fatty acids in ruminant diets did not substantially affect the impacts of the suckler-beef production system. Simultaneous application of several compatible practices can substantially mitigate the impacts of the suckler-beef production system. The application of certain practices (e.g. reducing ungrazed grass losses, fattening heifers not used for replacement and reducing calving age) reduced land occupation. Alternative uses for the "released land", e.g. the introduction of forest to sequester C into biomass, seems promising.

For dairy production systems, the assessment focused on a grass-based vs. maize-silage-based system, dual-purpose breed (Normande) vs. specialised breed (Holstein) and the effect of increasing milk yield per cow, using the ALCA approach. Independent of co-product handling methods, the impacts per kg of milk were lower with the maize-silage-based system and with Holstein cows (except for eutrophication). Increasing milk yield per cow by increasing feed energy intake and applying more intensive management (first calving at 2 years) decreased the impacts of milk and its beef co-product. The consequences of converting a maize-silage-based to a grass-based dairy farm in France to meet the increased domestic preference for grass-based milk were assessed using the CLCA approach. This farm conversion caused land-use change outside the dairy farm and thus substantially increased the impacts of the whole production system and the milk it produced.

Keywords: suckler-beef, dairy, production systems, attributional life cycle assessment (ALCA), consequential LCA (CLCA), land-use change, environmental impacts

Résumé

Analyse de cycle de vie de la production bovine : exploration de pratiques et de changements de système pour réduire les impacts environnementaux

Cette thèse porte sur l'étude des impacts environnementaux de systèmes de production de bovins. Le premier objectif était d'analyser et de comparer les impacts environnementaux de systèmes de production de viande et de lait par analyse de cycle de vie (ACV) attributionnelle. Les effets de pratiques d'atténuation de ces impacts ont été évalués pour les systèmes de production de viande. Le second objectif était un développement méthodologique afin d'explorer les conséquences possibles d'une préférence accrue pour un lait produit à base d'herbe, par ACV conséquentielle.

Dans un système de production de viande par le troupeau allaitant, le méthane entérique a été le principal contributeur à l'impact changement climatique, et la production de l'herbe a été la principale contributrice aux autres impacts (demande énergétique cumulée, eutrophisation, acidification, occupation du sol). L'atelier naisseur (vaches allaitantes et leurs veaux, génisses) a contribué de manière majeure aux impacts du système allaitant dans son ensemble. La pratique d'atténuation la plus efficace pour le système a été la diminution de l'âge au vêlage de 3 à 2 ans. L'utilisation de lipides riches en acides gras oméga-3 dans le régime a très peu affecté les impacts du système. L'application simultanée de plusieurs pratiques d'atténuation compatibles entre elles réduit sensiblement les impacts. L'application de pratiques telles que la réduction du gaspillage d'herbe, l'engraissement des génisses non utilisées pour le renouvellement et la diminution de l'âge au vêlage réduisent l'occupation du sol. Un usage alternatif des terres libérées tel que la plantation de forêt pour séquestrer du carbone dans la biomasse semble prometteur.

L'étude de systèmes de production de lait a été centrée sur les comparaisons de systèmes à base d'herbe ou d'ensilage de maïs, d'une race spécialisée (Holstein) ou mixte (Normande) et sur l'effet du niveau de production laitière par ACV attributionnelle. Quelle que soit la méthode d'attribution des impacts aux co-produits, les impacts par kg de lait ont été plus faibles pour les systèmes à base d'ensilage de maïs et pour les Holstein, sauf pour l'eutrophisation. L'accroissement de la production de lait par vache grâce à une consommation d'énergie accrue et au vêlage à 2 ans a permis de réduire les impacts du lait et de son co-produit viande. Les conséquences de la conversion d'une exploitation laitière utilisant beaucoup de maïs ensilage vers une exploitation utilisant de l'herbe comme unique source de fourrage pour répondre à une demande de lait produit à base d'herbe en France ont été évaluées par ACV conséquentielle. Cette conversion entraîne des changements notables de l'utilisation des sols en dehors de l'exploitation, et donc un fort accroissement des impacts du système dans son ensemble et du lait produit.

Mots-clés : troupeau allaitant, troupeau laitier, systèmes de production, analyse de cycle de vie (ACV) attributionnelle, ACV conséquentielle, changement d'usage des sols, impacts environnementaux

List of publications

1. International scientific journals

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Nguyen TTH, van der Werf HMG, Eugène M, Veysset P, Devun J, Chesneau G, Doreau M (2012) Effects of type of ration and allocation methods on the environmental impacts of beef-production systems. ***Livestock Science 145:239-251*** *(Chapter 2)*

Nguyen TTH, Doreau M, Eugène M, Corson MS, Garcia-Launay F, Chesneau G, van der Werf HMG. Effect of farming practices for greenhouse gas mitigation and subsequent alternative land-use on environmental impacts of beef-cattle production systems. ***Animal, accepted 3 September, 2012*** *(Chapter 3)*

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Nguyen TTH, Corson MS, Doreau M, Eugène M, van der Werf HMG. Consequential LCA of switching from maize-silage-based to grass-based dairy systems. **In preparation for *International Journal of Life cycle assessment*** *(Chapter 5)*

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Nguyen TTH, Doreau M, Eugène M, Corson MS, Garcia-Launay F, Chesneau G, van der Werf HMG, "Effect of farming practices and alternative land uses on greenhouse gas emissions of beef production systems", **63rd Annual Meeting EAAP**, Bratislava, Slovakia, August 27-31, 2012, Book of abstracts, p 339

Nguyen TTH, van der Werf HMG, Eugène M, Veysset P, Devun J, Chesneau G, Doreau M, "Effect of the enrichment of ruminant rations with omega 3 fatty acids on the environmental impacts of beef production systems", Proc. **8th International Symposium on the Nutrition of Herbivores**, Aberystwyth, UK, September 6-9, 2011, p 268

Doreau M, **Nguyen TTH**, van der Werf HMG, Martin C "Role of the nature of forages on methane emission in cattle", **63rd Annual Meeting EAAP**, Bratislava, Slovakia, August 27-31, 2012, Book of abstracts, p 295

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4. Posters

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Nguyen TTH, van der Werf HMG, Eugène M, Veysset P, Devun J, Chesneau G, Doreau M, « L'effet de l'enrichissement des rations en oméga 3 sur les impacts environnementaux des systèmes de production de viande bovine », **18è Journées de Rencontres Recherches Ruminants**, Paris, France, 7-8 décembre 2011, p 166

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Nguyen TTH, van der Werf HMG, Ponchant P, Bouvarel I, « Utiliser des contraintes environnementales pour formuler des aliments volailles à faible impact », **9è Journées de la Recherche Avicole**, Tours, France, 29-30 mars 2011, p 63

Nguyen TTH, Bouvarel I, Ponchant P, van der Werf HMG, "Using environmental constraints to formulate low-impact poultry feeds". Proc. **7th International Conference on Life Cycle Assessment in the Agri-Food sector**, Bari, Italy, September 22-24, 2010, pp 203 - 208

Chapter 1

General Introduction

1.1. Background

1.1.1. Growth of the livestock sector

World agriculture is facing a great challenge due to rapid growth of world population, the increase in human consumption and the increasing demand for bioenergy. Global population is estimated to grow annually by 76 million and to exceed 9 billion by 2050 (UN, 2009). In developing and emerging countries, food consumption, in particular animal-product consumption, has rapidly increased over the past decades because of economic growth, higher disposable incomes and urbanisation (Steinfeld et al., 2006). Since the 1980s, the growth of per capita animal-product consumption is higher than that of other major groups of food commodities (cereals, roots and tubers) in developing countries (FAO, 2009). In developing countries the per capita consumption of milk, meat and eggs has increased by factors of two, more than three and five, respectively, since the 1960s (FAO, 2009). In particular in China, per capita milk, meat and egg consumption has increased by a factor of ten, four and eight, respectively, over the same period (FAO, 2009). Global production of meat and milk is projected to double by 2050 in developing countries (Steinfeld et al., 2006). World agricultural production needs to increase by 60% to satisfy future global food demand over the next 40 years (OECD-FAO, 2012).

Competition for land between food, feed and biofuel production has become critical due to the limited ability to increase world arable land area (by less than 5% by 2050) (OECD-FAO, 2012). The use of biofuels to reduce dependence on fossil fuels has substantial effects on agricultural production. Biofuels constitute only 1% of the total energy consumption of the global transport sector, 94% of which is supplied by petroleum (FAO, 2008). It has been projected that the share of biofuels in total transport energy will increase to 2.3% in 2015 and 3.2% in 2030 (IEA, 2007). The increase in global production of biofuels is expected to consume a growing share of global crop production. This corresponds to an increase in world arable-land occupation from 1% in 2004 to 2.5-4.2% in 2030, depending on the scenario (FAO, 2008).

1.1.2. Environmental impacts of livestock

The livestock production sector is the fastest growing part of the agricultural economy (FAO, 2009). The livestock sector also has a major impact on global climate change, water and soil pollution, degradation of ecosystems, natural resources and biodiversity (Steinfeld et al., 2006). The awareness of the sector's contribution to greenhouse gas (GHG) emissions has been rapidly rising in recent years at the global, regional and national levels (Gerber et al., 2010). At the global scale, using a life cycle assessment (LCA) approach, livestock accounts for nearly 80% of agricultural emissions and 18% of total anthropogenic GHG emissions (Steinfeld et al., 2006). The world dairy sector (including milk and beef co-products) is estimated to contribute 4% of global anthropogenic GHG emissions (FAO, 2010). Using an LCA approach, the livestock sector is estimated to account for 85% of GHG emissions of European Union (EU) agriculture (Leip et al., 2010). Beef production and cow-milk production are the most important contributors (28-29% and 28-30%, respectively) to the total GHG emissions of the EU livestock sector (Weiss and Leip, 2012).

While the LCA approach considers activities across the world, a different approach per activity sector is used for national emission inventories, as required by the United Nations (UN). The French inventory agency reported that in 2009 agricultural activities in France contributed 20.9% of total French emissions (i.e., 10.0% for crops, 9.3% for livestock and 1.6% for other agricultural sources; CITEPA, 2011). The national inventory approach divides livestock emissions into three parts: direct animal emissions, crops for feed and on-farm energy emissions. Carbon dioxide (CO_2) emissions from fossil fuel used in the transport sector and other off-farm emissions are excluded. In addition, this estimate does not include emissions related to imported inputs, such as the production of some fertilisers (e.g. phosphate extraction) and several feed ingredients (mainly soybean) and their transport. Moreover, emissions related to land-use change outside France (e.g. deforestation associated with soybean) are not taken into account.

World livestock contributes substantially to GHG emissions from global agricultural activities and total anthropogenic emissions of nitrous oxide (N_2O) (75-80% and 65%, respectively) and methane (CH_4) (80% and 35-40%, respectively) (Steinfeld et al., 2006). At the global level, the most important sources of N_2O and CH_4 emissions from livestock are manure and enteric fermentation (Steinfeld et al., 2006). Methane emissions account for 52% of GHG emissions of the global dairy sector, while N_2O and CO_2 account for 27-38% and 21-10%, respectively (FAO, 2010). Nitrous oxide and CH_4 account for 18-24% and 21-29% of GHG emissions of EU livestock activities, respectively (Weiss and Leip, 2012). Beef and cow-milk production contribute substantially to the emissions of N_2O and CH_4 from EU livestock (60% and 81%, respectively) (Weiss and Leip, 2012). In France, livestock contribute substantially to national emissions of CH_4 (79%) and, to a lower extent, to those of N_2O (9.2%) (excluding land use and land-use change, LULUC) (CITEPA, 2010).

The growing awareness of the contribution of the livestock sector to environmental impacts, in particular climate change, results in increasing pressure from international and national policy makers to mitigate impacts. The EU has committed to mitigate environmental problems from agriculture (including livestock) since the 1991 Nitrate Directive, which recommended development of national policies to reduce nitrate pollution, especially by limiting manure N application to 170 kg/ha. This directive has indirect impacts on GHG mitigation, as it leads to lower N_2O emissions. Other EU directives that affect agricultural GHG emissions are the 2001 Directive on national emission ceilings for atmospheric pollutants, such as ammonia (NH_3, with an indirect effect on N_2O emissions); the 2009 Directive promoting 20% of renewable energy in total energy consumption by 2020; the 2010 EU-2020 Strategy to increase energy efficiency and reduce GHG emissions through national programs (that may include agriculture); and the 2010 Roadmap for moving to a low-carbon economy by 2050, with an objective to reduce GHG emissions from agriculture by 42-49% compared to 1990.

In France, the Nitrate Directive has been implemented in the second pillar (i.e. environmental) of the EU Common Agricultural Policy (CAP) by providing agro-environmental subsidies to reduce the use of mineral fertilisers, helping to conserve natural resources (phosphate), reduce N and P pollution and indirectly mitigate GHG emissions. National climate plans for France and other countries have been developed to propose mitigation strategies and their implementation. For the livestock sector, French national plans developed in 2011 aim to move farms towards grass-based systems, with an increase in legume crops, introduction of hedges to increase C sequestration, a decrease in tillage and an increase in energy-use efficiency. In general, policy efforts focus on urgent local problems, such as impacts of nitrate-related water pollution and reduction in the GHG emissions included in the national inventory.

1.2. Emission mitigation strategies

Many strategies have been proposed for mitigating impacts of the livestock sector; however, they focus only on GHG mitigation, in particular for CH_4 and N_2O, due to policy priorities.

The options for enteric CH_4 mitigation are well-known and regularly updated (Beauchemin et al., 2009; Martin et al., 2010; Cottle et al, 2011; Doreau et al, 2011a). Nutritional strategies for decreasing enteric CH_4 emissions are the most advanced and ready to be implemented (Martin et al., 2010). Among dietary strategies to mitigate enteric CH_4, the most efficient are increasing the proportion of cereal-rich concentrated feeds ("concentrates") in the diet and lipid supplementation (Doreau and Dollé, 2011).

1.2.1. Methane

Methane losses for diets containing up to 30-40% concentrates are about 6-7% of gross energy intake but only 2-3% for diets containing 80-90% concentrates (Martin et al., 2010). A reduction in enteric CH_4 is observed when the diet contains more than 50% concentrates (Sauvant and Giger-Reverdin, 2007) and is larger with higher intake (Sauvant and Giger-Reverdin, 2009). The effect of lipid enrichment in decreasing CH_4 emission is well established but highly variable. Giger-Reverdin et al. (2003) reported a mean decrease in enteric CH_4 emission by 2.3% for each percentage of lipid (on a dry matter (DM) basis) added to the dairy cow diet, independent of the type of fatty acid (FA) supplied. This was confirmed by Eugène et al. (2008), who reported a 2.3% reduction in dairy cow CH_4 emission (g/day) per percentage of lipid (on a DM basis) added, but no reduction was observed when corrected for intake (g/kg DM intake). Beauchemin et al. (2008), based on a meta-analysis of 17 studies on cattle and sheep, reported a higher reduction factor of lipid supplement on enteric CH_4 emission (5.6% decrease per percentage increase in lipid). However, Martin et al. (2010), analysing 28 publications on cattle, sheep and lambs, reported a mean reduction in enteric CH_4 emission of 3.8% per percentage of lipid added. The enrichment of diets with polyunsaturated FA such as linoleic acid (from soybean and sunflower) and linolenic acid (from linseed) has larger effects on enteric CH_4 emission (4.1% and 4.8% decrease per percentage of lipid added, respectively) (Martin et al., 2010). The reduction in enteric CH_4 by linseed supplement tends to have a long-term effect in dairy cows (Martin et al., 2008) and fattened bulls (Eugène et al., 2011).

The use of biofuel by-products such as distillers' dried grains seems a promising way to reduce CH_4 emissions (Beauchemin et al., 2009). Feeding biofuel by-products to ruminants improves efficiency of the biofuel industry and reduces competition for crops (as food or feed). The use of additives such as ionophores, organic acids and plant extracts (e.g.

condensed tannins, saponins, essentials oils) is considered a potential way to reduce CH_4 emissions, but their effectiveness is unclear, especially in the long term, and more information about their possible side-effects must be known. The use of ionophores is not always effective, and their effect may be limited to the short term (a few months). Organic acids must be used in large amounts in the diet and are expensive. Tannins, either as extracts or in specific plants, reduce CH_4 emission per kg DM intake, but negative consequences on di et digestibility and thus on a nimal performances are often a problem. The effect of most saponin-rich plants is minor; tea saponins may be effective but more research is needed.

Other strategies to decrease enteric CH_4 emission through biotechnologies such as immunisation, biological control, probiotics, and elimination of protozoa have been explored, but more research is needed to evaluate their effectiveness (Martin et al., 2010). Increasing animal productivity is suggested as one way to reduce enteric CH_4 emission per unit of animal product, because it decreases the ratio of energy for maintenance to energy for production (Beauchemin et al., 2009; Doreau and Dollé, 2011). However, for dairy cows, the effects of increasing animal productivity can be partially compensated by decreasing reproductive performance and number of lactations per lifetime (Garnsworthy, 2004).

Manure CH_4 emission is related to indigestible faecal organic matter and manure management systems and treatment. The IPCC (2006) guidelines show large variability in manure CH_4-emission factors. Some factors influencing manure CH_4 emissions, such as temperature are difficult to control, but solutions such as aeration, which decreases anaerobic conditions, and reducing the time of storage before application have been effective. In addition, moderate acidification of slurry has been found to decrease its CH_4 emission (Petersen et al., 2012). Combustion of CH_4 using a flare system has been suggested, as has anaerobic digestion for producing biogas. These latter techniques are not easy to implement, and their economic viability has been questioned.

1.2.2. Nitrous oxide

Options for decreasing N_2O emissions from agriculture have been widely studied. Nitrous oxide emissions are related to manure management (from excretion to manure application) and mineral N management. The efficiency of N transfer from the soil to animals is an important key to decrease N_2O emissions. The most attractive approaches are those that improve N-use efficiency of animals and crops, i.e. reducing N input without reducing yields (Schils et al., 2012). Reduction in crude protein intake by using low-protein feed supplements, such as maize silage, result in less N excreted in urine (Kebreab et al., 2001; Luo et al., 2008). The use of condensed tannin in ruminant diets increases the fraction of N excreted in manure (compared to urine) and thus lowers N_2O emissions, as emissions from manure N are lower than those from urine N (Eckard et al., 2010; Schils et al., 2012).

Reducing grazing time (usually accompanied by increased increasing maize silage in the diet) is considered an effective way to reduce N_2O emissions (Schils et al., 2007 and 2012), as is increased use of leguminous crops, which do not require N fertilisation. Avoiding grazing on wet soil can reduce N_2O emissions and nitrate leaching, avoid soil damage and reduce the deposition of animal excreta on wet soil (Ledgard et al., 2006; Luo et al., 2008). N_2O emissions can also be decreased by increasing anaerobic conditions during manure storage and treatment (Schils et al., 2012). Improved manure and mineral fertiliser application can improve the use of N inputs by plants and pasture and hence reduce N losses. The rate, source, timing and placement of mineral N fertiliser or manure are important factors that affect N_2O emissions (Eckard et al., 2010). Urease and nitrification inhibitors can be used as practices to decrease N_2O emissions (Luo et al., 2010), but further research is needed to confirm their on-farm effectiveness over periods exceeding one year (Eckard et al., 2010). Apart from technical limitations, one major drawback of these techniques is their relatively high cost.

1.2.3. Mitigation strategies for whole farm systems

Mitigation strategies are usually proposed for individual GHGs (e.g. in the UN Framework Convention on Climate Change recommendations) or to solve a specific environmental problem, without investigating interaction among GHGs or other environmental impacts. Measures that decrease emission of one gas can be counterbalanced by an unwanted increase in emissions of other gases (e.g. CH_4, CO_2, NH_3) or in other impacts (e.g. energy use, eutrophication). Manure management affects GHG emissions because NH_3 volatilisation is an important indirect source of N_2O (Petersen and Sommer, 2011). Decreased grazing on wet soil to reduce N_2O emissions may increase NH_3 emissions from animals in housing (Luo et al., 2010). Anaerobic conditions to decrease N_2O emissions during manure management can increase CH_4 emissions (Schils et al., 2012).

It has been shown that the use of a high-concentrate (86% of DM) diet for finishing young bulls strongly decreased enteric CH_4 emissions (56% lower per day) compared to a diet based on maize silage (37% concentrates) or hay (51% concentrates) (Doreau et al., 2011b). This diet did not decrease total GHG emissions of the entire bull fattening phase when expressed per day, but did decrease them by 21% when expressed per kg of live weight gain, despite increased emissions of N_2O and CO_2 from production of the additional concentrates used (Doreau et al., 2011b). Other impacts, however, such as energy use and eutrophication, were highest for the high-concentrate diet (Nguyen et al., 2012).

Ledgard et al. (2009), using a systems approach, reported that grass/clover pastures without fertiliser can have lower GHG emissions and energy use, with no difference in nutrient losses to water, compared to grass pastures fertilised with 200 kg N/year. Therefore, when assessing the effectiveness of mitigation strategies, a systems approach is necessary to consider possible interactions and verify that the strategy does not increase emissions elsewhere in the production chain (Martin et al., 2010; Eckard et al., 2010; Schils et al., 2012).

The LCA approach is appropriate for assessing environmental impacts of a production system in a global and integrated vision of the entire production chain, using a variety of impact categories and functional units. Whole-farm simulation modelling has been developed to investigate the effectiveness of mitigation options (Crosson et al., 2011), but these approaches are usually limited to GHG emissions (Casey and Holden, 2005; Weiske et al., 2006; Beukes et al., 2006; Beauchemin et al. 2011) and sometimes only focus on on-farm emissions, without integrating the associated upstream processes such as N inputs and production of purchased feed (Schils et al., 2007).

1.3. Life cycle assessment of cattle production systems: State-of-the-art

Life cycle assessment is a method for assessing environmental impacts of a product, process or service by including all phases of its life cycle (ISO, 2006; JRC, 2010). LCA was initially developed to assess environmental impacts of industrial sectors and production processes and has been applied in agriculture for crop production since the 1990s (Audsley et al., 1997) and for milk production from 2000 (Cederberg and Mattsson, 2000, Haas et al., 2001).

In recent years, LCA studies on livestock production have rapidly increased in number, in particular after the FAO report *Livestock's Long Shadow* (Steinfeld et al., 2006), which was the first study investigating environmental impacts of global livestock production using an LCA approach. This report highlighted negative impacts of livestock production, in particular GHG emissions and other environmental impacts, which are predicted to increase with the inevitable increase in livestock production in the future. The report stressed the high contributions of livestock production to global climate change. In cattle production, most studies have focused on dairy production and GHG emissions and to a lesser extent on suckler-cattle beef production and other environmental impacts.

In France, efforts have been focused on bot h dairy and suckler-beef production systems because of the large proportion of suckler cows in the cattle French herd and its contribution to the French economy. Based on i ts farm network, the French Livestock Institute has performed LCA studies of the main production systems from farm-level data (Gac et al., 2010; Dollé et al., 2011). At INRA, LCA studies for cattle production have been focused on specific production systems, using data from farm surveys (van der Werf et al., 2009, Veysset et al., 2010, 2011) or from experiments associated with expert knowledge of farming practices (Doreau et al., 2011b; Nguyen et al., 2012).

LCA has become an internationally accepted method, used widely in the agricultural sector to assess environmental impacts and identify hotspots (Thomassen et al., 2008b). A hotspot is defined as an element that has a high contribution to the environmental impacts of a product (Guinée et al., 2002). Although the conceptual framework of LCA is well defined by ISO normalisation (ISO, 2006), LCA studies vary widely in their implementation and methodologies.

1.3.1. Principles of life cycle assessment

LCA addresses potential environmental impacts (e.g. resource use, environmental consequences of emissions) throughout a product's life cycle, from raw material acquisition through production, use, end-of-life treatment, recycling and final disposal (i.e. cradle-to-grave). LCA consists of four main phases: goal and scope definition, inventory analysis, impact assessment and interpretation (ISO, 2006).

The first step of LCA is the definition of the goal and scope of the study. The former includes the objectives, intended audience and application, while the latter includes definition of the system boundary, functional unit(s), type of modelling framework (attributional or consequential, described later) and method for co-product handling (ISO, 2006; JRC, 2010). In the second step, life cycle inventory (LCI) analysis, input/output data are acquired and the system (i.e., product life cycle) is modelled to quantify emissions and resource use throughout the product's life cycle (JRC, 2010). In the third step, life cycle impact assessment (LCIA), input and output flows in the inventory are converted into impact indicator values to estimate potential environmental impacts of the product's life cycle (JRC, 2010). In the final phase, life cycle interpretation, the LCI or LCIA results (or both) are summarised and discussed as a basis for conclusions, recommendations and decision-making in accordance with the study's goal and scope (ISO, 2006). The value and robustness of all results, choices and assumptions are also evaluated.

1.3.2. Attributional and consequential LCA modelling

Attributional LCA (ALCA) accounts only for immediate physical flows (i.e., resources, material, energy and emissions) involved in the life cycle of a product, while consequential LCA (CLCA) additionally describes how qualitative or quantitative changes in these flows occur, in response to changes in the life cycle of a product (Ekvall and Weidema, 2004). The modelling principles of ALCA and CLCA are the same, but the primary difference between them is the processes included in the system (Zamagni et al., 2012). ALCA typically uses average data for each process within the life cycle, while CLCA includes only the process affected by a change in the life cycle. CLCA modelling includes market mechanisms and tends to integrate economic models (Earles and Halog, 2011). Thus, CLCA does not reflect an existing (or predicted) specific or average supply-chain but reflects possible future environmental impacts from a change in the life cycle of the product under study (e.g. due to an increase in production).

Affected processes (i.e. marginal processes)

The definition of the system boundary (system delimitation) determines which unit processes will be included in the LCA (ISO, 2006). In CLCA, the processes/technologies included are only those actually affected by the change in production (Weidema, 2003), which are referred to as "marginal". A step-wise procedure for identifying marginal processes/technologies described by Ekvall and Weidema (2004) includes:

1. What are the relevant time aspects?

   This step identifies the temporal scope of the study, which is concerned with the short-term and long-term effects of decisions. A short-term effect includes only effects on the use of existing production capacity, and a short-term study may have a temporal scope of about 5 years (Mathiesen et al., 2009). A long-term effect includes adaptation of the production capacity to the change, and the use of this capacity is assumed to be constant.

2. Are specific processes or the overall market affected?

   This step identifies the properties, position and relevant market segment of the products in question, which makes it possible to define competing products on the market segments affected.

3 What is the trend in the market?

   This step identifies the overall trend in demand in the relevant market segment. If the demand decreases at a higher rate than investments in replacement of the existing capacity, the marginal technology, which is the most likely to be phased out, is the technology with the highest short-term costs. If demand is increasing or decreasing at a slower rate, the long-term marginal technology is likely to be the technology chosen when new production capacity is installed.

4. What technologies are flexible?

   If the production capacity of a technology is fixed, its capacity cannot be affected by any decisions based on LCA results and cannot be the long-term marginal technology. If the production volume is fixed, it cannot even be the short-term marginal technology. Technologies may be constrained due to technical, natural (e.g. milk from natural grassland), political (e.g. emission limits, quotas), or market-related (e.g. for co-products such as milk and beef) constraints. However, a change in demand (e.g. electricity) may influence political constraints (e.g. on nuclear power). The effects of changes on the constraints should be taken into account. Hence, it is difficult to model these changes; it is a simplification to consider that constraints are treated as fixed, even in studies for long-term decision support.

5. What technology is actually affected?

The marginal technology is among the technologies on the market capable of responding to changes in demand. When market trends of the demand are increasing or constant, the marginal technology is identified as the most competitive technology. Conversely, in situations with a decreasing market trend, the marginal technology is identified as the least competitive technology. The most or least competitive technology can be determined primarily on the basis of the price relations between the technologies (Schmidt and Weidema, 2008). Technologies that are not likely to respond to a change in demand should not be included in a CLCA since this will not reflect the actual change in environmental impact (Weidema, 2003).

Method for co-product handling

ALCA and CLCA also differ in the way in which they handle co-products. According to ISO (2006), within ALCA, co-products should be handled following the stepwise procedure:

Step 1: whenever possible, allocation should be avoided by (1) dividing the unit process to be allocated into two or more sub-processes and collecting the input and output data related to these sub-processes, or (2) expanding the product system to include the additional functions related to the co-products.

Step 2: where allocation cannot be avoided, the inputs and outputs of the system should be partitioned between its different products or functions in a way that reflects the underlying physical relationship between them; i.e. they should reflect the way in which the quantitative changes of the products or functions delivered by the system change the other inputs and outputs.

Step 3: where physical relationship alone cannot be established or used as the basis for allocation, the inputs should be allocated between the products and functions in a way that reflects other relationships between them; e.g. based on the economic value of the products.

System expansion means that the boundary of the system investigated is expanded to include the alternative production of co-products. The identified alternative product must exist, and substitution of the co-product of studied system by the alternative product must be relevant and actually occur in the market (Jolliet et al., 2010). In the case of a dairy production system, co-products of milk are culled dairy cows and calves. It is necessary to identify the alternative products for these co-products. Indeed, avoiding allocation by using system expansion to handle co-products is optional within ALCA, while co-product allocation is used most (Thomassen et al., 2008a). However, within CLCA, the only way to deal with co-products is using system expansion because it reflects the consequences of a change in demand (Weidema, 2003).

CLCA studies applied to agriculture have most frequently concerned (oil) crops and biofuel production (Dalgaard et al., 2008; Schmidt, 2008; Reinhard and Zah, 2009; Silalertruksa et al., 2009; Smyth and Murphy, 2011). To our knowledge, the only study using CLCA modelling for ruminant production system is by Thomassen et al. (2008a), who modelled consequences of meeting increased demand for milk with an additional dairy farm. Affected processes included electricity produced by a natural gas power plant and marginal barley and soybean meal (i.e. marginal feed energy and protein supplies, respectively) (Thomassen et al., 2008a). As for co-product handling, system expansion was applied by replacing meat from the additional dairy system with a mix of suckler-beef and pork (Thomassen et al., 2008a).

1.3.3. Variability of implementation and methodology among LCA studies applied for dairy and beef-cattle production systems

Goal and scope definition

The goal of the study may vary widely to meet specific scientific or policy demands and the intended use of the study. The goals of LCA studies in ruminant production usually consist of (1) quantifying environmental impacts (mostly GHG emissions only) of the production system, comparing impacts of production modes (e.g. organic *vs*. conventional systems, high *vs*. low inputs) or illustrating the variability in impacts among different farms; or (2) identifying environmental hotspots and possible improvement options. Sometimes the goal is to assess the effectiveness of mitigation options for the whole production system (Schils et al., 2005; Weiske et al., 2006; Stewart et al., 2009; Beukes et al., 2010; del Prado et al., 2010; Beauchemin et al., 2011). The goal often focuses on LCA methodology itself, e.g. comparison of the influence of co-product handling method on estimated impacts of milk and beef co-products (Cederberg and Stadig, 2003; Flysjö et al., 2011a; Zehetmeier et al., 2012; Flysjö et al., 2012), implementation of CLCA (Thomassen et al., 2008a), uncertainty (Basset-Mens et al., 2009; Flysjö et al., 2011b; Henriksson et al., 2011), and consideration of LULUC effects (Cederberg et al., 2011; Vellinga and Hoving, 2011; Flysjö et al., 2012). The goal can also be to quantify environmental impacts of dairy and/or beef cattle production at the national, European or global level (Steinfeld et al., 2006; Capper et al., 2009; FAO, 2010; Weiss and Leip, 2012; Cederberg et al., 2012).

Ideally, the system boundary of an LCA is cradle-to-grave, but LCAs of livestock systems (or agriculture in general) often set it as cradle-to-farm-gate, in which processes after the farm-gate (e.g. slaughtering, food processing, retailing, consumption and waste handling) are excluded. Nonetheless, some studies also included slaughtering (Weiss and Leip, 2012) or primary meat processing (Peters et al., 2010). The FAO (2010) and Cederberg et al. (2012) studies went even further, including food processing and transport to the retailer (i.e. cradle-to-retailer). Only Heller and Keoleian (2011) assessed the impacts of organic milk from cradle-to-grave (i.e. to waste disposal). In contrast, some studies focus on one physiological phase, i.e. the fattening phase in beef-cattle production systems (Ogino et al., 2004; Doreau et al., 2011b; Nguyen et al., 2012), reflecting the fact that beef-cattle fattening is often done on specialised farms. Most studies excluded the construction and use of buildings, machinery and medicines due to their relatively small contribution or a lack of data.

Choice of functional unit(s)

The functional unit represents the main function of a production system by which all physical flows (i.e. resources, material, energy and emissions) involved in the life cycle of a product are quantified. The functional unit allows comparison of alternative production systems, and only products with similar function can be compared (ISO, 2006). The choice of the functional unit depends on the goal and context of the study. The primary function of farming systems is to provide quantities of food, feed or other forms of biomass to the market; thus, the most common functional unit is based on mass, usually "kg of product". Functional units used by ruminant-production LCAs vary widely, despite being mass-based: "kg milk", "kg energy corrected milk", "kg fat and protein corrected milk", "kg live weight", "kg carcass weight", "kg (free bone) meat". Some studies express results per "kg of live weight gain" (Doreau et al., 2011b; Nguyen et al., 2012) or per "one marketed animal" (Ogino et al., 2004 and 2007) when the system is limited to one part of animal's life. It is difficult to compare studies using different functional units without a way to standardise them (Basset-Mens et al., 2009; van der Werf et al., 2009). Some studies express results per "kg of protein" because the primary function of animal products is to provide humans with animal protein, and this unit allows comparison of different animal production sectors and products (Stewart et al., 2009; de Vries and de Boer, 2010; FAO, 2010). The environmental impacts of farming systems have also been expressed per (on-farm and off-farm) unit of area (Casey and Holden, 2005; Basset-Mens et al., 2009; van der Werf et al., 2009; Foley et al., 2011) or per on-farm unit of grassland area (Haas et al., 2001). The use of area as functional unit can be seen as a way to reflect ecosystem services supplied by farming systems, may be more appropriate to assess regional impacts such as eutrophication, and allows analysis of effects of changes in production intensity.

Co-product handling

The most commonly applied method for co-product handling of inputs (in particular feed ingredients) is economic allocation (de Vries and de Boer, 2010) because production is primarily driven by the economic value of products (Jolliet et al., 2010). For inputs, some studies also use mass allocation (Cederberg and Mattsson, 2000; Thomassen et al., 2008a; O'Brien et al., 2012), energy-based allocation (Pelletier et al., 2010), nitrogen-based allocation (Weiss and Leip, 2012) or system expansion (Thomassen et al., 2008a; Nguyen et al., 2010). As for outputs, none of the studies of suckler-beef production systems differentiated the impacts of different types of animals produced (e.g. cull cows, bulls, finished heifers). In dairy production systems, impacts were allocated to milk and beef based on biophysical allocation (i.e. allocation based on f eed-energy requirements needed to produce milk and animals) (Cederberg and Mattsson, 2000; Basset-Mens et al., 2009; IDF, 2010; O'Brien et al., 2012), economic allocation (Casey and Holden, 2005; Thomassen et al., 2008b; van der Werf et al., 2009), or protein allocation (FAO, 2010; Weiss and Leip, 2012). Several studies specifically compared several allocation methods and system expansion for milk and meat co-products (Cederberg and Stadig, 2003; Thomassen et al., 2008a; Flysjö et al., 2011a; Kristensen et al., 2011).

Inventory analysis

The sources of farm-level data used in LCA studies vary widely. Some studies used real-farm data obtained from surveys, either analysing all surveyed farms individually (Cederberg and Mattsson, 2000; Haas et al., 2001; Casey and Holden, 2006; Williams et al., 2006; Thomassen et al., 2008b; van der Werf et al., 2009; Peters et al., 2010; Vayssieres et al., 2010; Kristensen et al., 2011) or defining representative farm types from the data for analysis (Gac et al., 2010; Dollé et al., 2011; Veysset et al., 2010 a nd 2011). Some studies used experimental data (Schils et al., 2005; Lovett et al., 2006, 2008; O'Brien et al., 2010, 2012). Some studies used regional and national statistics to model representative production systems for the region under investigation (Phetteplace et al., 2001; Capper et al., 2009; Pelletier et al., 2010; White et al., 2010; Flysjö et al., 2011a, b; Henriksson et al., 2011; Cederberg et al., 2012). A combination of regional and national statistics and experimental data was used in Basset-Mens et al. (2009) and Foley et al. (2011). Weiss and Leip (2012) used the CAPRI database, mostly based on regional and national statistics, for the EU agricultural sector, whereas the FAO (2010) required a large amount of data from sources such as global statistics (FAOSTAT), satellite data on gross primary production, and the literature.

Sources of emission factors are highly variable in LCA studies. Emission factors used to estimate GHG emissions are primary taken from IPCC guidelines (Tier 1 and 2). Some studies used published emission factors which were considered more relevant for the systems under investigation (Haas et al., 2001; Ogino et al., 2004, 2007; Williams et al., 2006; Thomassen et al., 2008b). Other studies used emission factors derived from IPCC methodology for national inventories, which can be considered IPCC Tier 3 (Basset-Mens et al., 2009; Beauchemin et al., 2010; Beukes et al., 2010; Peters et al., 2010; White et al., 2010; Veysset et al., 2010, 20 11). Some studies used both literature and IPCC emission factors (Casey and Holden, 2005, 2006; Beauchemin et al., 2011; Flysjö et al., 2011b; Henriksson et al., 2011; Cederberg et al., 2012; O'Brien et al., 2012). Only Doreau et al. (2011b) used measured values for enteric CH_4 emissions and IPCC emission factors for other GHGs. Some studies took into account C sequestration in grassland (Schils et al., 2005; Pelletier et al., 2010; Doreau et al., 2011b; Veysset et al., 2010, 2011), whereas others considered C emissions or sinks in pasture due to changes in management practices (related to land use (LU)) (Stewart et al., 2009; Rotz et al., 2010; Beauchemin et al., 2011; Vellinga and Hoving, 2011). The effects of global land-use change (LUC) were taken into account in Nguyen et al. (2010) and Weiss and Leip (2012).

Simulation models have been used in livestock LCA studies, but most focus on GHG emissions. A whole-farm GHG model was used by Olesen et al. (2006), Weiske et al. (2006), Beauchemin et al. (2010, 2011), Rotz et al., 2010 and del Prado et al. (2010). Other studies coupled several models to achieve a whole-farm GHG analysis (Lovett et al., 2006, 2008; Beukes et al., 2010; O'Brien et al., 2010; White et al., 2010; Vellinga and Hoving, 2011; Veysset et al., 2010, 2011). Weiss and Leip (2012) used a livestock-production module of an agricultural sector economic model covering EU-27 and Norway.

Impact assessment

The most common impact assessed in LCA studies is climate change (i.e. GHG emissions), including LUC or not. Many studies also assess impacts such as (non-renewable) energy use and potential acidification and eutrophication (Haas et al., 2001; Cederberg and Mattsson, 2000; Cederberg and Stadig, 2003; Ogino et al., 2004, 2007; Williams et al., 2006; Thomassen et al., 2008a, b; van der Werf et al.; 2009, Basset-Mens et al., 2009; Nguyen et al., 2010; Pelletier et al., 2010; Vayssieres et al., 2010; O'Brien et al., 2012). White et al. (2010) and Peters et al. (2010) evaluated the risk of nitrate leaching and soil erosion potential, respectively. These reflect that the major environmental concern at both global and regional levels is climate change and, to a lesser extent, eutrophication (Yan et al., 2011).

Some studies assessed impacts on land use, but only as the area of land (on-farm and/or off-farm) occupied. Human toxicity and ecotoxicity impacts related to the accumulation of heavy metals and pesticide use are rarely assessed due to a lack of inventory data and toxicity characterisation factors. In fact, many LCA studies aggregate the impact of heavy metals and pesticide use to assess toxicity, but as these substances differ very much with respect to persistence in the environment (months for pesticides, centuries of heavy metals) it is difficult to choose an appropriate temporal horizon to assess toxicity impacts of these pollutants. Only Cederberg and Mattsson (2000) and Cederberg and Stadig (2003) listed pesticides used and quantified total active substances per functional unit, while van der Werf et al. (2009) reported a terrestrial ecotoxicity impact (based only on heavy metal emissions). Only Haas et al. (2001) included biodiversity and animal welfare as impact categories.

1.4. Objectives of the thesis

1.4.1. Scope of the thesis

The milk and beef sectors represented 34% and 20% of the total economic output of the EU livestock sector in 2007, respectively (Eurostat, 2008). According to economic output, France ranked first among EU-country beef sectors and second among EU-country milk sectors (after Germany) (Eurostat, 2008). In France, the milk- and beef-production sectors have similar economic value and herd size. The economic contribution of milk and beef sectors to French livestock production was highest (31 and 34%, respectively), followed by the poultry and pig sectors (13 and 12%, respectively) (Eurostat, 2008). Due to milk quotas, the French dairy cow herd strongly decreased from 7.5 million in 1970 to 4.5 million in 1992 and 3.8 million in 2006 (Institut de l'Elevage, 2011). At the same time, the French suckler-cow herd substantially increased, from 2.3 million in 1970 to 4.2 million in 2006 (Institut de l'Elevage, 2011). Thus, in 2010, 35% of beef production in France came from dairy cattle and 65% from suckler cattle (Institut de l'Elevage, 2011). French dairy and suckler-beef production systems have highly diverse feeding strategies and farming practices (Devun and Guinot, 2012; Réseaux d'élevage Charolais, 2009). Therefore, this thesis focuses on both suckler-beef and dairy production systems, due to their similar importance (in economic and production volumes) and their diversity (of feeding systems and farming practices).

1.4.2. Main objective: Analysis and comparison of production systems

More LCA studies have focused on dairy production systems than on suckler-beef production systems, and even fewer have focused on suckler-beef production in France. Pelletier et al. (2010) reported that the suckler cow-calf herd was the largest contributor to impacts of the entire suckler-beef production system. The suckler cow-calf herd is mainly based on grassland production; hence, differences between production modes of this herd may be small.

First, this thesis quantifies environmental impacts of a representative suckler-beef production system in France, identifies the contribution of the suckler cow-calf and fattening herds, and then assesses effects of several feeding strategies for the fattening herd. Second, this representative suckler-beef production system is used as a baseline to evaluate the effects of GHG mitigation strategies, most of which are applied to the cow-calf herd due to its relatively large impact in the system. Alternative land-use options are explored on land "released" from agricultural use when mitigation practices allow production of the same amount of animal products on less land.

In France, dairy systems are based either on grass or both grass and maize silage. One of the most important issues for dairy production is to know the implications of the relative environmental impacts of grass and maize silage. The grass-based system is considered to be more environmentally friendly than a system based on both grass and maize silage according to the CAP. In France, subsidies arising in part from the second pillar of the CAP support grass-based systems. A change in the feeding system will induce a change in the farm system and in farming practices. Another issue in dairy production, which affects suckler-beef production, is the use of dual-purpose breeds. Therefore, in this thesis, the feeding system and cow breed are considered important factors.

Supplementation of omega-3 FA in ruminant diets is considered one of the most effective strategies to decrease enteric CH_4 emissions (Martin et al., 2010). In addition, it can improve milk and meat quality by increasing omega-3 FA content in animal products (Doreau et al., 2011c). However, it is necessary to assess the effects of the enrichment of ruminant diets with omega-3 FA on emissions of other GHGs and on other environmental impacts of the entire production system.

1.4.3. Second objective: Methodology development

Several methodological developments are explored in the thesis. Suckler-beef production systems produce several types of animals, such as finished cull cows, finished heifers, fattened bulls, and breeding bulls. These animals are raised with different production practices and have different economic values, but the influence of these differences on their environmental impacts has not been analysed to date. This thesis therefore investigates impacts of different animal co-products with several allocation methods.

LCA is sometimes criticised because it only assesses negative effects on the environment. However, suckler cow-calf herds in France, based mainly on grassland in hilly or mountainous regions, contribute to rural development and preserve the existing landscape and its biodiversity. These additional functions of the suckler cow-calf herd are recognised and promoted through agro-environmental measures of the second pillar of the CAP supporting grass-based systems. Therefore, as an example of taking environmental services into account, these positive effects are considered in the thesis (via economic allocation) as one function of these farming systems.

In dairy production, replacement of silage-maize fields with grass pastures is promoted to reduce nitrate leaching, soil erosion and increase biodiversity of farming systems. The use of grass also reduces feed cost and sensitivity of dairy farms to increased input prices. This raises the question of the environmental consequences of converting a grass and maize-silage based dairy farm to exclusively a grass-based dairy farm. To answer this question, the CLCA approach is used, including the effect on global LULUC.

1.5. Outline

The thesis consists of six chapters (Figure 1). Chapter 1 (this one) is the general introduction of the thesis. Chapters 2-5 have the format of scientific papers. Chapters 2 and 3 focus on beef-cattle production systems, whereas chapters 4 and 5 focus on dairy production systems. In chapter 2, environmental impacts of beef-cattle production systems using different rations are assessed. Several methods for allocating impacts to different types of animals based on live weight mass, protein mass and economic values are compared, and economic allocation considering agro-environmental subsidies is performed. Chapter 3 assesses the environmental impacts of farming practices aiming to reduce GHG emissions from beef-cattle production. Alternative land-use is assessed when permanent grassland becomes available due to more efficient farming practices. In chapter 4, six dairy farms differentiated by the proportions of grass and maize silage as forage for the herd and by cow breed (Holstein vs. Normande) are compared. Different methods of co-product handling (i.e. biophysical, protein, economic allocation and system expansion) are compared. Chapters 2-4 are based on the ALCA approach whereas chapter 5 is based on the CLCA approach. Chapter 5 analyses environmental consequences of meeting an increase in preference for grass-based milk by converting a maize-silage-based dairy farm to a grass-based dairy farm. Finally, chapter 6 provides a general discussion and conclusions of the work.

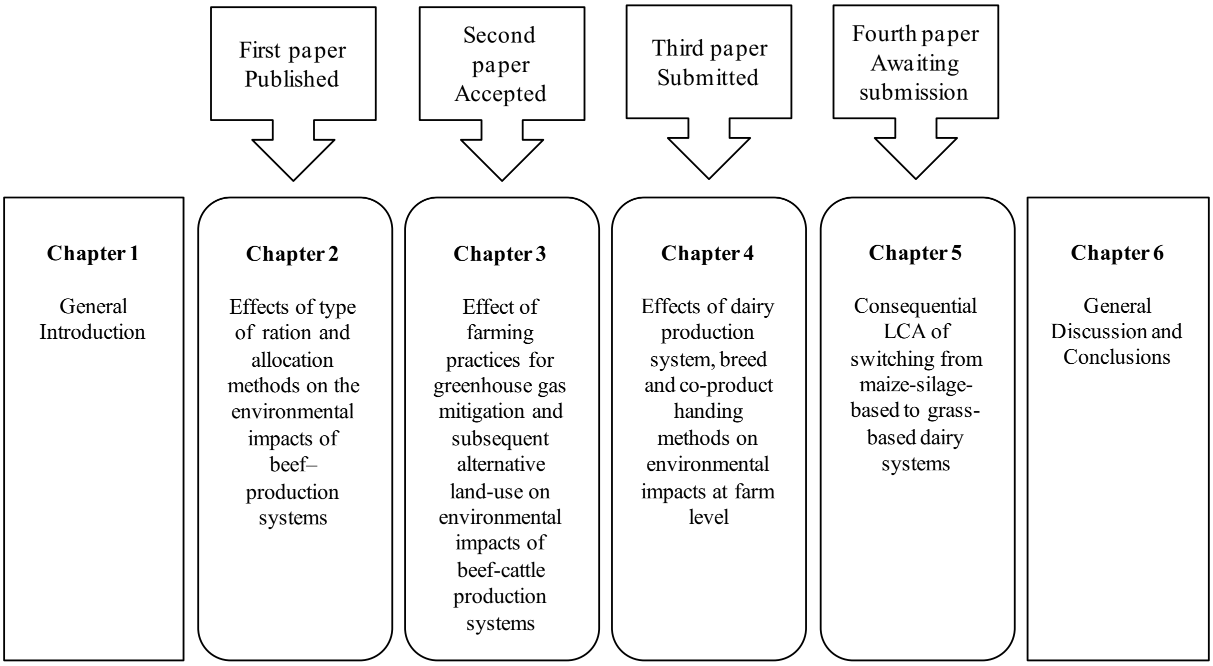

Figure 1: Diagram of the thesis framework

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Chapter 2

Effects of type of ration and allocation methods on the environmental impacts of beef-production systems

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Effects of type of ration and allocation methods on the environmental impacts of beef-production systems

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Abstract

Four complete beef-production systems consisting each of two stages were compared. The systems were formed by combining two diets for the cow-calf herd with finishing heifers stage -- St (Standard) and O3 (maximising omega-3 fatty acids (FAs) using wrapped grass silage) -- with four diets for the bull-fattening herd stage-- SM (silage maize starch), SML (silage maize starch plus linseed, rich in omega-3 FAs), FC (fibre-based concentrate), and SCL (starch-based concentrate plus linseed): St-SM, O3-SML, St-FC and O3-SCL. Life Cycle Assessments applied to these systems (from cradle to farm gate for a one-year period) estimated that their environmental impacts, per kg of carcass mass, ranged from 27.0-27.9 kg CO_2 equivalents (eq), 64.8-73.4 MJ, 94-98 g PO_4^{3-} eq, 168-173 g SO_2 eq, 47-48 m^2a for climate change (CC, not including effect of land use and land-use change, LULUC), cumulative energy demand (CED), eutrophication potential, acidification potential and land occupation, respectively. Consideration of LULUC decreased CC from 8 to 10%. Minor impact differences between these systems were observed, except for CED of St-FC, mainly because more energy was needed to dehydrate beet pulp and lucerne. CC of O3-SCL was 3% lower than CC of St-SM. Most of the environmental impacts of beef-production systems originated from the cow-calf herd with finishing heifers (73-97%), which indicates that research on the reduction of environmental impacts of this type of beef-production system should focus on this herd. For the cow-calf herd with finishing heifers, comparison of several allocation methods revealed that allocation method strongly affected the impacts per kg of carcass mass of the breeding bull and finished cull cows and, to a much lesser extent, those of fattened bulls and finished heifers. Consideration of both products (several animal types) and the ecosystem services supplied by these systems seems a promising perspective. This concept needs to be discussed and developed as an approach to consider the multi-functionality of farming systems.

Keywords: beef, feeding strategy, omega-3 supplementation, life cycle assessment, allocation method, environmental services

1. Introduction

Worldwide the livestock sector was estimated to contribute 18% of global greenhouse gas (GHG) emissions, according to a Life Cycle Assessment (LCA) approach (Steinfeld *et al.*, 2006). Methane (CH_4) is the most significant (58-63%) contributor to GHG emissions from beef systems (Veysset *et al.*, 2010). Supplementation of diets with lipids is one of the most effective strategies for reducing enteric CH_4 emissions by ruminants (Beauchemin *et al.*, 2009). Martin *et al.* (2008) reported that feeding lipids rich in omega-3 fatty acids (FAs) from linseed significantly decreased enteric CH_4 emissions from dairy cows. Enteric CH_4 production by bulls fed a high-concentrate diet based on cereals supplemented with extruded linseed was reduced by 23% (g/kg live weight gain) in comparison with a high-concentrate diet based on fibre-rich co-products (Eugène *et al.*, 2011). However, CH_4 mitigation strategies must be assessed in a global vision of production systems to evaluate all GHG emissions and other environmental impacts (Martin *et al.*, 2010).

In France, production systems for beef cows are based on grass, but fattening systems are diversified. For a same type of production, e.g. young bulls, there are several drivers for choosing a feeding system. The first one is the type of forage (based on grass or maize silage) and the proportion of forage relative to concentrate feed (Nguyen et al., 2012). The second one is the nature of concentrates. Beef quality is a major consumer concern. A primary target in improving meat's nutritional quality is increasing its concentration of n-3 polyunsaturated fatty acids (FA). Indeed these FA play a role in the reduction of the risk of coronary heart disease and in infant development (Doreau *et al.*, 2011a). Beef products can be enriched naturally with omega-3 FAs through provision of feed rich in linolenic acid, such as linseed, fresh grass or wrapped grass silage. Independently of meat quality, another strategy for beef fattening is the use of by-products rich in fibre, which avoids food competition with humans by reducing the use of cereals for animal feeding.

The main objective of this study was to investigate the environmental impacts, using a LCA approach, of a standard beef-production system in France by comparing two systems, one based on feed rich in omega-3 FA and one with co-products rich in fibre. These beef production systems corresponded to a grassland-suckler cow-calf herd with finishing heifers and a bull-fattening herd. Grassland-based production systems contribute to sustainable rural development due to the ecosystem services they provide: landscape quality, biodiversity and carbon (C) sequestration. An additional objective was to analyse different allocation methods used to attribute environmental impacts to the co-products delivered by production systems. The choice of allocation method has generated much discussion in LCA studies on dairy systems regarding the co-products of milk and meat. In our beef-production systems, co-products were the types of meats from fattened bulls, finished heifers, finished cull cows, and a breeding bull. Ecosystem services supplied by these grassland-based production systems can also be considered as a co-product; we will explore this option.

2. Materials and methods

2.1. Goal definition

The goal of this study was to investigate four beef-production strategies practiced in France, two of which produce omega-3 FA-enriched beef. These systems were characterised according to ration strategies for the suckler cow-calf herd with finishing heifers and the bull-fattening herd. We analysed the effect of different allocation methods, such as economic allocation (including the provision of ecosystem services or not), mass allocation, and allocation based on protein content, on potential environmental impacts for each co-product delivered by the system.

2.2. Scope definition

Description of French beef production systems

Each of the four production systems (Fig. 1) consists of two herds. The suckler cow-calf herd with finishing heifers (to be designated as *cow-calf herd in the rest of this paper*) produces weaned male calves or pre-fattened bulls, finished heifers, finished cull cows and a breeding bull. The weaned male calves or pre-fattened bulls are transferred to the bull-fattening herd, which yields fattened bulls. The systems are based on the Charolais breed as it represents 40% of the French suckler-cow herd (Institut de l'Elevage, 2010). Two production methods were compared for the cow-calf herd. The first was the standard (St) cow-calf herd, which is most frequently practiced in the Charolais basin. The second, the omega-3 (O3) cow-calf herd, aimed to maximise the animals' omega-3 FA intake by using wrapped grass silage, which can be easily adopted by farmers. Four production methods were compared for the bull-fattening herd. The first was a standard bull-fattening herd using a diet rich in starch based on silage maize (SM). The second was a bull-fattening herd using a diet rich in starch (based on silage maize) supplemented with linseeds (SML). The third was a bull-fattening herd using a fibre-based concentrate diet (FC). The last used a starch-based concentrate supplemented with a linseed diet (SCL). We combined the two herds to study the following four beef-production systems: 1) St-SM 2) St-FC, 3) O3-SML, and 4) O3-SCL. All rations were formulated to satisfy beef-cattle nutrient requirements according to animal characteristics and feed-composition values, based on recommendations of INRA beef researchers and data tables (INRA, 2007). Details for both phases of the four systems are given below.

As suckler-cow farming practices in the Charolais basin are highly diverse, our systems were modelled based on “Charolais Beef Cattle Farm Networks” of the French Livestock Institute (Réseaux d’Elevage Charolais, 2009) in consultation with beef researchers and experts. Both St and O3 cow-calf herds consist of 70 cows that annually provide 62 weaned calves (Table 1). The replacement rate, defined as the proportion of heifers replacing cull cows, was 23%. The cow-calf herd consists of four components for St and three components for O3. The first is reproduction; its output consists of weaned male calves, weaned female calves not used for replacement cull cows, cull cows and a breeding bull. The second component is rearing female calves from weaning (9 months) to finishing at 33 months. The third component is the finishing of cull cows, i.e. fattened before sending to the slaughterhouse. The last component (only for St) is pre-fattening of male calves for 2 months after weaning. The St and O3 cow-calf herd systems were built to reflect two types of actual farming practices which differ with respect to the calving period and the age at which male calves are sent to the bull-fattening herd. In the St, herd calves are born in February and weaned at 9 months, and male calves are pre-fattened for 2 months (reaching 430 kg live weight) with concentrate feed (20% crude protein) and hay before passing to the bull-fattening herd. In the O3, herd calves are born in January and weaned at 9 months, and male calves (350 kg live weight) are sent directly to the bull-fattening herd.

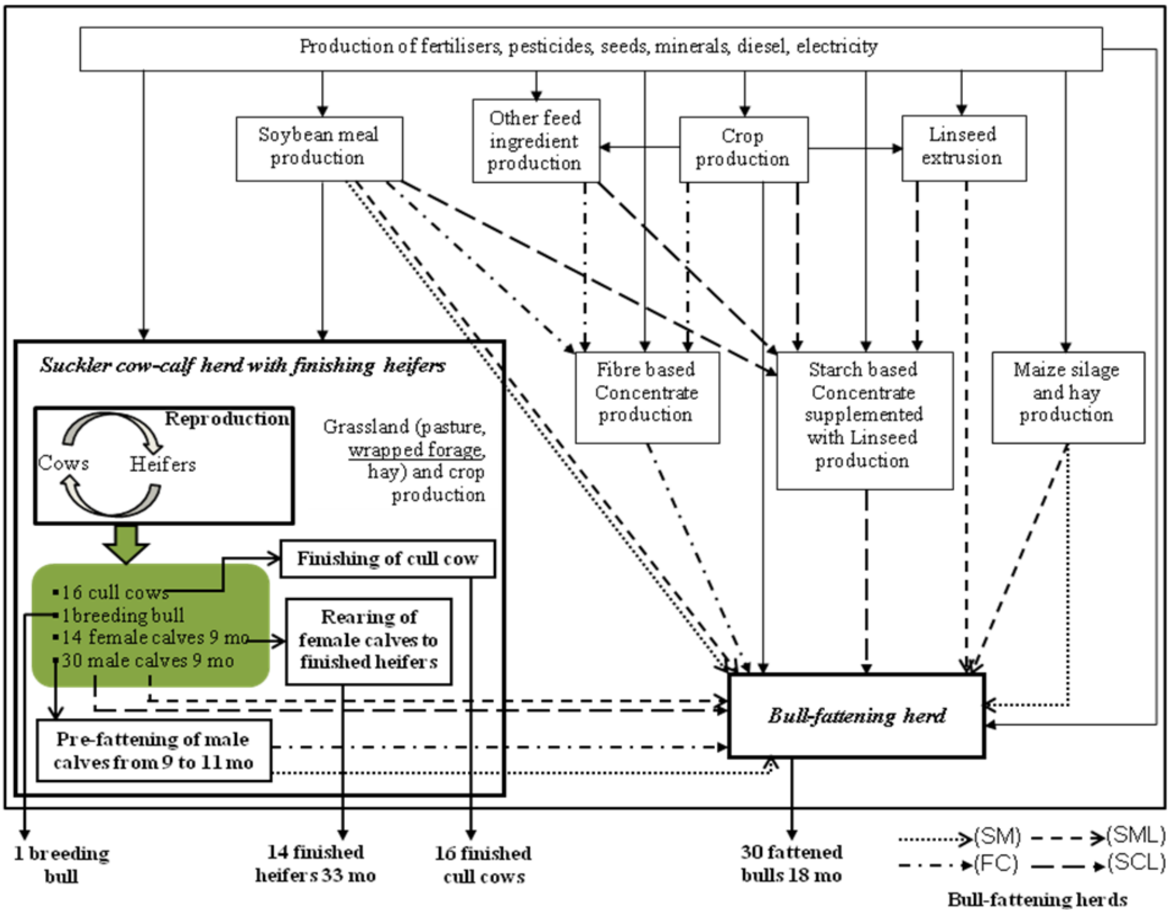

Figure 1: "Cradle to farm-gate" life cycle of the four beef-production systems

St: Standard suckler cow-calf herd with finishing heifers

O3: Suckler cow-calf herd with finishing heifers enriched in omega-3 FAs through pasture and wrapped grass silage

SM: Standard bull-fattening herd using a diet rich in starch based on maize silage

SML: Bull-fattening herd using a diet rich in starch (based on maize silage) supplemented with linseeds

FC: Bull-fattening herd using a fibre-based concentrate diet

SCL: Bull-fattening herd using a starch-based concentrate supplemented with linseeds

Underline: Indicates this aspect is only present in O3 suckler cow-calf herd with finishing heifers

Pre-fattening of male calves from 9 to 11 mo. is only present in St suckler cow-calf herd with finishing heifers

Allocation methods were applied for these animals of the common phase, i.e. the reproduction component, before passing to the other periods

Table 1: Inputs and outputs of the four beef production systems

Inputs (t dry matter)	**St-SM**	**O3-SML**	**St-FC**	**O3-SCL**
Feed for cow-calf herd with finishing heifer	**St**	**O3**	**St**	**O3**
Pastured grass	276.5	270.8	276.5	270.8
Hay	175.4	82.8	175.4	82.8
Wrapped grass silage	-	91.8	-	91.8
Cereals	76.5	66.9	76.5	66.9
Mix meal*	6.8	4.8	6.8	4.8
Feed for bull-fattening herd	**SM**	**SML**	**FC**	**SCL**
Maize silage	31.7	38.5	-	-
Wheat	15.0	13.0	-	-
Soybean meal	9.2	10.3	-	-
Croquelin®	-	5.9		
Fibre-based concentrate***	-	-	53.3	-
Starch-lipid-based concentrate***	-	-	-	58.7
Others	2.1	2.6	7.9	8.5
Animal outputs	**St-SM**	**O3-SML**	**St-FC**	**O3-SCL**
	Number of animal-kg live weight per animal			
Breeding bull	1-990	1-990	1-990	1-990
Finished cull cows	16-798	16-802	16-798	16-802
Finished heifers	14-695	14-701	14-695	14-701
Fattened bulls	30-720	30-720	30-720	30-720

For feed management, both St and O3 are situated in the grassland zone of the Charolais basin and are classified as extensive systems with 1.2 livestock units per ha of forage area and 7.5 months grazing from April to November. One livestock unit is defined as an animal that consumes 5t DM/year (Gac et al., 2010a). Indoors in winter, the St herd is fed with hay and concentrates (mainly based on cereals produced on-farm and a mix meal which consisted of 30% soybean meal, 40% rapeseed meal and 30% sunflower meal) produced off-farm, whereas the O3 herd is fed with wrapped grass silage, hay and concentrates (cereals and mix meal). Wrapped grass silage, i.e. grass silage at 55% dry matter (DM) covered by plastic, has a higher omega-3 FA content than hay (Arrigo, 2010). Cull cows are finished for 100 days with a concentrate diet and hay (St herd) or wrapped grass silage (O3 herd). Weaned female calves which are not destined to be used for replacement cows are reared as heifers to be used for replacement until 29 months and then finished at pasture supplemented with cereals over 4 months to produce finished heifers.

Pre-fattened bulls from St are finished in the SM and FC bull-fattening herds. Weaned male calves from O3 are finished in the SML and SCL bull-fattening herds. The SM herd was located in the Pays de la Loire region (western France), which is a cereal-producing region. This phase was modelled based on "Typical Case: Young bull-fattening in Pays de la Loire" of the farm networks of the French Livestock Institute (Sarzeaud *et al.,* 2009). The pre-fattened male calves are fed a high-forage diet composed of 58% maize silage, 24% wheat, 15% soybean meal, 2% hay, and 1% minerals (DM basis), resulting in an average daily live weight gain (ADG) of 1.40 kg/d. The SML herd, also located in the Pays de la Loire region, is modelled on the SM herd with a portion of the wheat replaced by extruded linseed. The diet is composed of 58% maize silage, 17% wheat, 14% soybean meal, 8% Croquelin® (containing 50% extruded linseed, 30% wheat bran and 20% sunflower meal, Valorex, Combourtillé, France), 2% hay, and 1% minerals (DM basis). We assumed that animals in the SML herd are provided the same quantity of net energy for growth (i.e. 63 MJ/d) as those in the SM herd. Since lipid supplementation is known to improve beef cattle performance (Clinquart *et al.*, 1995) we assumed that the ADG of the SML herd (1.6% lipid added) animals is 5% higher than that of animals in the SM herd.

The other two diets are high in concentrates, and have been chosen because they represent two different options. One of them (FC) is rich in fibrous by-products; the interest is to use less cereals (which can feed humans) and to minimize the risk of digestive health problems such as acidosis. The other one (SML) is rich in cereals, and maximises the net energy value of the diet, by addition of lipids. The FC herd is located in the Champagne-Ardenne region (northern France), where cattle are frequently fed beet pulp and dehydrated lucerne. The diet (DM basis) of the FC herd consists of 13% straw and 87% concentrate including 22% wheat bran, 22% dehydrated lucerne and 21% dehydrated beet pulp (Eugène *et al.*, 2011). We assumed that animals in FC herd are provided 63 MJ/d of net energy for growth resulting in an ADG of 1.62 kg/d (unpublished experimental data, Mialon M.M., pers. comm.). The SCL herd is located in the Aquitaine region (south-western France), where high-concentrate diets based on cereals are frequently used to fatten bulls. The SCL diet (DM basis) consists of 13% barley straw and 87% concentrate rich in starch and lipids that includes 46% cereals and 6% extruded linseed (Eugène *et al.*, 2011) that provided 62 MJ/d of net energy for growth resulting in an ADG of 1.71 kg/d (unpublished experimental data, Mialon M.M., pers. comm.).

The carcass yields of breeding bulls, finished heifers and finished cull cows were 57%, 56% and 54%, respectively, according to expert knowledge and the slaughterhouse database of the INRA Herbivore Research Unit. The carcass yield of fattened bulls was 59% according to

Institut de l'Elevage (2011) and expert knowledge. All cereals produced on farms with cow-calf herds are consumed on the farm by the herd. Annual ration plans for cow-calf herds and bull-fattening herds and animal outputs of the four systems are presented in Table 1 and in supporting information Table S1 and Table S2.

System boundary and delimitations

This is a cradle-to-farm-gate study for a one-year period, i.e. the studied system includes the production and delivery of inputs used for grassland and cereals produced on-farm, of feed produced off-farm, herd management and associated upstream processes, emissions from the animals and manure storage. The application of manure for cereals and pasture is included, as are buildings. The transport and slaughter of animals leaving the system are not included. Veterinary medicines are not included because of lack of data.

Functional unit and allocation of co-products

The functional units were 1 kg of carcass mass at the farm exit gate for the whole systems, 1 kg live weight gain for each herd and 1 ha of land occupied (both for the whole system and each herd). Carcass mass produced was calculated by multiplying animal live weight at the farm gate by the specific carcass yields for each animal type. Economic allocation was used for feed ingredients resulting from processes yielding several co-products. Allocation was applied for animals delivered from the reproduction component of the cow-calf herd (a breeding bull, cull cows, weaned female calves not used for replacement and weaned male calves). We compared different methods for the allocation of impacts to co-products:

1. Allocation on live weight mass. This implies that there is no difference in quality between live weight mass of different animal types. All live weight mass delivered from the reproduction component carried the same environmental burden.
2. Allocation based on protein mass. This was based on the protein content in the live weight mass (CORPEN, 2001) of each co-product delivered from the reproduction component of cow-calf herd.

3. Economic allocation. This was based on the market value of the live weight mass of each co-product delivered from the reproduction component. The prices per kg of live weight mass for each co-product were based on data from the French Livestock Institute for the 2004-2007 period (Réseaux d'Elevage Charolais, 2004, 2005, 2006, 2007).

4. Economic allocation with agro-environmental subsidies. Agricultural activity, and in particular grassland-based production systems, has multiple functions such as food production, renewable natural-resource management, landscape and biodiversity conservation and contribution to the socio-economic viability of rural areas (Renting et al., 2009). The agro-environmental measures of the European Union's Common Agricultural Policy (CAP) encourage farmers to maintain the environmental functions of agriculture. Thus, we attributed the environmental impacts of the studied system to these two functions. We used economic allocation based on beef product income as specified above and on agro-environmental subsidies for grassland according to the "Second Pillar" of the 2003 CAP reform in French conditions, to attribute environmental impacts to beef products (per kg of live weight mass) and to environmental services (per hectare of grassland). Subsidies or financial incentives vary between EU countries, and with time, therefore this calculation should be considered as an example for taking into account the effect of public policies on the environmental impact. Allocation techniques are summarised in Table 2.

Table 2: Outputs from two reproduction components of Standard and Omega-3 FA-enriched suckler cow-calf herds with finishing heifers and allocation factors following different allocation techniques

	Standard suckler cow-calf herds with finishing heifers					Omega-3 FA-enriched suckler cow-calf herds with finishing heifers				
	Breeding bull	Cull cow	Weaned female calf	Weaned male calf	Grassland	Breeding bull	Cull cow	Weaned female calf	Weaned male calf	Grassland
Number of animals or grassland area (ha)	1	16	14	30	82	1	16	14	30	81
Live weight mass of animals (kg)	990	690	300	350	-	990	690	300	350	-
Average price of animal products 2004-2007 (€/kg of live weight) or grassland subsidy (€/ha of grassland)	1.3	1.7	2.2	2.6	70	1.3	1.7	2.2	2.6	70
Protein content in live weight mass (g/kg)	75	125	181	181	-	75	125	181	181	-
Allocation factors										
Mass allocation (%)	4	41	16	39	-	4	41	16	39	-
Allocation based on protein content (%)	2	33	19	46	-	2	33	19	46	-
Economic allocation between animal products (%)	2	32	18	48	-	2	32	18	48	-
Economic allocation between animal products and grassland subsidy (%)	2	29	16	44	9	2	29	16	44	9

2.3. Life cycle inventory analysis

Feed production

The cropping and grassland area was determined from total annual feed requirements for the beef production systems and the 4-year (2004-2007) average yields of pasture and crops based on the data of AGRESTE (2009). Grassland management was modelled on grassland production, the stocking rate of the production system and the amount of forage DM required for cattle in winter. The grassland area consisted of 88% permanent and 12% temporary pastures (AGRESTE, 2009). We assumed that permanent grassland did not require tilling and sowing operations. Permanent grassland had a yield of 5.6 t DM/ha/year, 23% of which was harvested as conserved forage (hay and/or wrapped grass silage). Temporary grassland had a higher yield (8.3 t DM/ha/year, 75% was harvested as conserved forage) and was renewed every 5 years by tillage and seeding. Grass not harvested as conserved forage was available for ingestion by animals during grazing. For several reasons (selective grazing, trampling of grass, unfavourable weather conditions) a part of the grass grown is not ingested, this “loss” corresponded to 31.5% of grass dry matter available for grazing. Losses during conservation for both hay and wrapped grass silage were assumed to be 6% of the initial DM. Apart from manure excreted on pasture during grazing, application rates of mineral and organic fertilisers were based on the data of Réseaux d’Elevage Charolais (2009) with 1.2 livestock units per ha of forage area as the stocking rate. Pesticide use and other farm practices for grassland (Table S3) were based on a recent survey of agricultural practices (AGRESTE, 2006).

The period considered for crops begins with soil preparation for the specific crop and ends with soil preparation for the next cash crop. This period may include a catch crop. Data on input use and crop management (Table S3) were based on a recent survey of agricultural practices (AGRESTE, 2006). Data for soybean production (70% soybean from central-western and 30% from southern Brazil) and transport in Brazil was based on Prudêncio da Silva *et al.* (2010).

Major feed ingredient production

We considered that the transformation of soybean into soybean meal and oil occurred in Brazil based on data by Nemecek and Kägi (2007) and Jungbluth *et al.* (2007). According to the main French producer of extruded linseeds (Valorex, pers. comm.), the extrusion process required 60 kWh of electricity and 0.21 kWh of natural gas to produce 1 t of Croquelin®. According to information provided by a French dehydration cooperative (Coop de France Déshydratation, pers. comm.), dehydration of lucerne and beet pulp from 25% to 90% DM required 6 GJ/t (mainly supplied by hard coal coke (59%), natural gas (27%) and light fuel oil (10%)) of dehydrated product. Delivery of feed ingredients to the farm and feed mill and the delivery of concentrate feed to the farm were included. We assumed that the fibre-rich concentrate was produced in the Champagne-Ardenne region and starch and lipid-rich concentrate in the Aquitaine region.

Buildings and operations

This study included the production and transportation of materials required for the construction of buildings such as cattle housing, forage and manure storage based on the GES'TIM guide (Gac *et al.*, 2010a). It was assumed that the cattle housing and manure storage had a 30-year life span and that the forage storage had a 50-year life span. However, energy use and emissions during the construction or disposal of the building were not included because of lack of information. The use of machines and energy for housing illumination, feeding, mulching, carrying manure out of housing and cleaning were included as farming operations, based on data from Dollé and Duyck (2007).

Emissions and effect of land use and land-use change (LULUC) on soil C balance

Enteric CH_4 emissions were estimated for each class of cattle according to the method developed by Vermorel *et al.* (2008) for cattle production in France and used for French gaseous-emissions inventories. This method uses animals' net-energy requirements, converted into metabolisable energy intake (MEI), and conversion factors from MEI to methane energy (Y'm=MJ CH_4/100 MJ MEI), to express CH_4 emissions per kg of DM intake (DMI). This allowed the consideration of diet characteristics for each class of cattle (Table 3). This method is not applicable to diets rich in lipids. To include the effect of diets supplemented with lipids rich in omega-3 FAs on ruminants' enteric methane production, a 4.8% reduction factor of enteric methane production (g CH_4/kg DMI) per percentage unit of added lipids was applied, based on results from a quantitative analysis (Martin *et al.*, 2010).

Table 3: Estimation of enteric methane (g/kg dry matter intake) produced by different types of animal in different periods in St-SM and St-FC beef-production systems

	Indoors	Grazing season		
		Late spring	Summer	Autumn
Multiparous cow	22.6	20.0	16.9	15.8
Primiparous cow	22.5	19.3	17.0	16.5
Heifer (>24 months)	21.5	22.3	18.5	-
Heifer (12-24 months)	23.5	19.8	18.1	17.1
Heifer (<12 months)	22.6	-	-	17.5
Breeding bull (>24 months)	21.6	19.1	16.5	15.2
Breeding bull (12-24 months)	23.1	20.9	18.2	17.0
Breeding bull (<12 months)	24.9	-	-	18.6
Pre-finisher	18.7	-	-	-
Cull cow	23.7	-	-	-
Growing heifer	-	-	18.7	17.6
Fattening bull with maize silage	25.1	-	-	-
Fattening bull with fibre-rich concentrate	20.3	-	-	-

St: Standard suckler cow-calf herd with finishing heifers

SM: Standard bull-fattening herd using a diet rich in starch based on maize silage

FC: Bull-fattening herd using a fibre-based concentrate diet

The cow-calf herd was housed in deep bedding from December to April (4.5 months). The manure accumulated indoors was removed once a year. For bull-fattening herds, it was assumed that cattle remained indoors during the fattening period and that slurry was evacuated and stored outside the animal housing without a natural crust cover. Methane, nitrous oxide and ammonia emissions from manure produced by cattle in housing and during storage were included as part of livestock manure management, and emissions from manure deposited during grazing were included as part of grassland production. Nitrogen excretion was calculated as the difference between the animal's total nitrogen intake in feed and the nitrogen retained for growth (meat production) for each grazing and indoor period. For P-excreted on pa sture, our estimation was based on C orpen (2001) taking into account the number of livestock units and the duration of grazing per ha of grassland. A summary of emission factors used for livestock, cropping and grassland production and their sources is presented in Table 4.

Table 4: Emissions sources, equation or emission factor used and reference

Pollutant/source		**Equation/emission factor**	**Reference**
Manure management			
Direct N_2O	= N excreted (kg) x EF[1] x 44/28		IPCC 2006 Tier 2
	deep bedding manure	EF = 0.07 kg N_2O-N/kg N	
	slurry without natural crust cover	EF = 0 kg N_2O-N/kg N	
Indirect N_2O	= N excreted (kg) x $Frac_{Gas}$[2] (%) x 0.01 x 44/28		IPCC 2006 Tier 2
	deep bedding manure	$Frac_{Gas}$ = 30%	
	slurry without natural crust cover	$Frac_{Gas}$ = 40%	
CH_4	= [[GEI[3] x (1-DE[4]%)/100 + UE*GEI[5]] x 0.92)/18.45] x 0.17 x 0.67 x MCF[6] (%)/100		IPCC 2006 Tier 2
	deep bedding manure	UE = 0.04; MCF = 4	
	slurry without natural crust cover	UE = 0.04 for SM and SML and 0.02 for SCL and FC; MCF = 27%	
NH_3	in housing	=0.12 x N excreted (kg) x 17/14	Payraudeau *et al.,* 2007
	in storage	=0.06 x N remaining (kg) x 17/14	
Cropping and grassland production			
Direct N_2O	=[[(mineral N (kg) + liquid N (kg) + cattle manure N (kg) + residue N (kg)] x 0.01 + N deposited by grazing x 0.02)] x 44/28		IPCC 2006 Tier 2
Indirect N_2O	=[[[(mineral N (kg) + liquid N (kg)] x 0.1 + cattle manure N (kg) x 0.2] x 0.01 +N-NO_3 (kg) x 0.0075] x 44/28		IPCC 2006 Tier 2
NO_x	= 0.21 x N_2O (kg)		Nemecek and Kägi, 2007
NH_3	= (0.02 x mineral N (kg) + 0.08 x liquid N (kg) + 0.076 x cattle manure N (kg) + 0.08 x N deposited by grazing) x 17/14		Nemecek and Kägi, 2007; Payraudeau *et al.,* 2007; CORPEN 2006
NO_3	Cropping		Basset-Mens *et al.*, 2007
	Grassland	= 8.77 $e^{0.003 \text{ x grazing days/ha/LU7}}$ x 62/14	Vertès *et al.,* 1997
P leaching	Cropping	=0.07 kg P/ (ha x yr)	Nemecek and Kägi, 2007
	Grassland	=0.06 kg P/ (ha x yr)	
P run-off	=P run-off lost x [1 + 0.2/80 x mineral P_2O_5 (kg) + 0.4/80 x manure P_2O_5 (kg) + 0.7/80 x P_2O_5 deposited by grazing (kg)]		Nemecek and Kägi, 2007
	Cropping	P run-off lost = 0.175 kg P/ (ha x yr)	
	Grassland	P run-off lost = 0.15 kg P/ (ha x yr)	
P erosion	=10000 x (80 x 0.033 x 0.38 x 0.65 x effect of the vegetation cover factor) x 0.00095 x 1.86 x 0.2 kg P/ (ha x yr)		Nemecek and Kägi, 2007; Nemecek *et al.*, 2003

[1]EF: emission factor for direct N_2O emissions from manure management

[2]$Frac_{Gac}$: % of managed manure nitrogen for production system that volatilises as NH_3 and NO_x

[3]GEI: gross energy intake

[4]DE: digestibility of the feed

[5]UE x GEI: urinary energy expressed as fraction of GEI

[6]MCF: methane conversion factor from each manure-management system (in %)

[7]LU: livestock unit

The effect of land use on C sequestration in grassland was estimated according to Dollé *et al.* (2009) from measurements of C in soils summarised by Arrouays *et al*. (2002). For permanent grassland, i.e. grasslands older than 30 years, C sequestration was estimated at 200 kg C/ha/year. We assumed that temporary grassland was maintained for five years and was followed by an annual crop for two years. C sequestration was assumed to equal 500 kg C/ha of temporary grassland/year and C release during the subsequent two years of annual crops was estimated at 1000 kg C/ha/year. As a result, there is a net C sequestration for temporary grassland of 100 kg C/ha/year. We assumed that other annual crop area was converted from permanent grassland more than 20 years ago and that agricultural practices for these crops had no effect on soil carbon. The part of Brazilian forest converted to soybean was estimated based on Prudêncio da Silva *et al.* (2010). In order to better conform to current practice with respect to the effect of land-use change on C release due to conversion of Brazilian forest to cropland we decided to adopt a value of 740 t CO_2/ha as recommended in PAS 2050 (2008) among others, instead of the value of 120 t CO_2/ha used in the Ecoinvent database (Jungbluth et al., 2007). Indeed, the latter estimate corresponds to the estimated of 20% of the above-ground biomass which is burnt, but the remaining 80% is ignored, the reason for this is not specified (Prudêncio da Silva, 2011).

2.4. Life cycle impact assessment

The impact categories considered were climate change (CC), eutrophication potential (EP), acidification potential (AP), cumulative energy demand (CED) and land occupation (LO). The indicator value for each impact category was determined by multiplying the aggregated resources used and the aggregated emissions of each individual substance with a characterisation factor for each impact category to which it may potentially contribute, as implemented in the Ecoinvent® v2.0 database. CC is defined as the potential impact of gaseous emissions on the heat radiation absorption in the atmosphere. It was calculated according to the 100-year global warming potential factors in kg CO_2 equivalent (eq), CH_4: 25, N_2O: 298, CO_2: 1 (IPCC, 2007). Climate change does not take into account the effect of LULUC on C sequestration in grassland and C release due to conversion of Brazilian forest to cropland, whereas CC/LULUC takes into account these effects. CED accounts for the use of renewable and non-renewable energy resources by using the conversion efficiencies of primary energy carriers. Eutrophication covers all potential impacts of high environmental levels of macronutrients, in particular N and P. EP was calculated using the generic EP factors in kg PO_4 eq, NH_3: 0.35, NO_3: 0.1, NO_2: 0.13, NO_x: 0.13, PO_4: 1 (Guinée et al., 2002). Acidifying pollutants have a wide variety of impacts on soil, groundwater, surface water, biological organisms, ecosystems and materials. AP was calculated using the average European AP factors in kg SO_2 eq, NH_3: 1.6, NO_2: 0.5, NO_x: 0.5, SO_2: 1.2 (Guinée et al., 2002). Land occupation, including on-farm and off-farm area, refers to the loss of land as a resource in the sense of being temporarily unavailable for other purposes due to crop and grass production.

3. Results

3.1. Environmental impacts

The environmental impacts of these systems are presented per kg of carcass mass and per ha of land occupied during a year (Table 5). Thus, carcass mass for each system consisted of fattened bulls, but also the corresponding output of the cow-calf herd (i.e. a breeding bull, finished heifers and finished cull cows, see Fig. 1). We observed minor differences between the four systems per kg of carcass mass and per ha (+/- 5% relative to St-SM) for all impact categories except CED. The lowest values per kg of carcass mass for CC and CC/LULUC were obtained in O3-SCL. The lowest values for CED were observed in St-SM. The highest CED values were observed in St-FC, with 13 and 17% per kg of carcass mass and per ha, respectively, higher than those for St-SM. Consideration of the effect of LULUC induced a reduction of 9% of the CC impact for both functional units.

Table 5: Impacts per kg of carcass mass and per ha of land occupation (both on-farm and off-farm) of the four beef-production systems

	St-SM	O3-SML	St-FC	O3-SCL	St-SM	O3-SML	St-FC	O3-SCL
	per kg of carcass mass				per ha of land occupation			
Climate change, kg CO_2 eq	27.8	27.7	27.9	27.0	5770	5880	5980	5780
Climate change/LULUC, kg CO_2 eq	25.5	25.5	25.3	24.4	5290	5400	5420	5240
Cumulative energy demand, MJ	64.8	68.4	73.4	71.1	13470	14510	15720	15260
Eutrophication, g PO_4^{3-} eq	0.098	0.098	0.094	0.098	20.5	20.9	20.1	21.1
Acidification, g SO_2 eq	0.169	0.173	0.168	0.173	35.2	36.7	35.9	37.1

St: Standard suckler cow-calf herd with finishing heifers
O3: Suckler cow-calf herd with finishing heifers enriched in omega-3 FAs through pasture and wrapped grass silage
SM: Standard bull-fattening herd using a diet rich in starch based on maize silage
SML: Bull-fattening herd using a diet rich in starch (based on maize silage) supplemented with linseeds
FC: Bull-fattening herd using a fibre-based concentrate diet
SCL: Bull-fattening herd using a starch-based concentrate supplemented with linseeds
LULUC: Land use and land-use change

In our systems, enteric fermentation was the greatest contributor (39-41%) to CC followed by grassland production (24-25%), emissions from manure management (21-22%), and production of other feed (9-10%). Both building and farming operation only contributed 4% to CC (Fig. 2). The contribution of grassland production to CC/LULUC was lower than it was to CC. For other impact categories, grassland production was the major contributor to the environmental impacts of production systems (58-63% of EP, 46-47% of AP and 81-83% of LO). The production of other feed contributed 19-23% to EP, 12-13% to AP and 14-16% to LO. For CED impact, grassland production, other feed production, building and farming operation contributed approximately a third each. The emissions from manure contributed 17-18% and 37-39% to EP and AP, respectively.

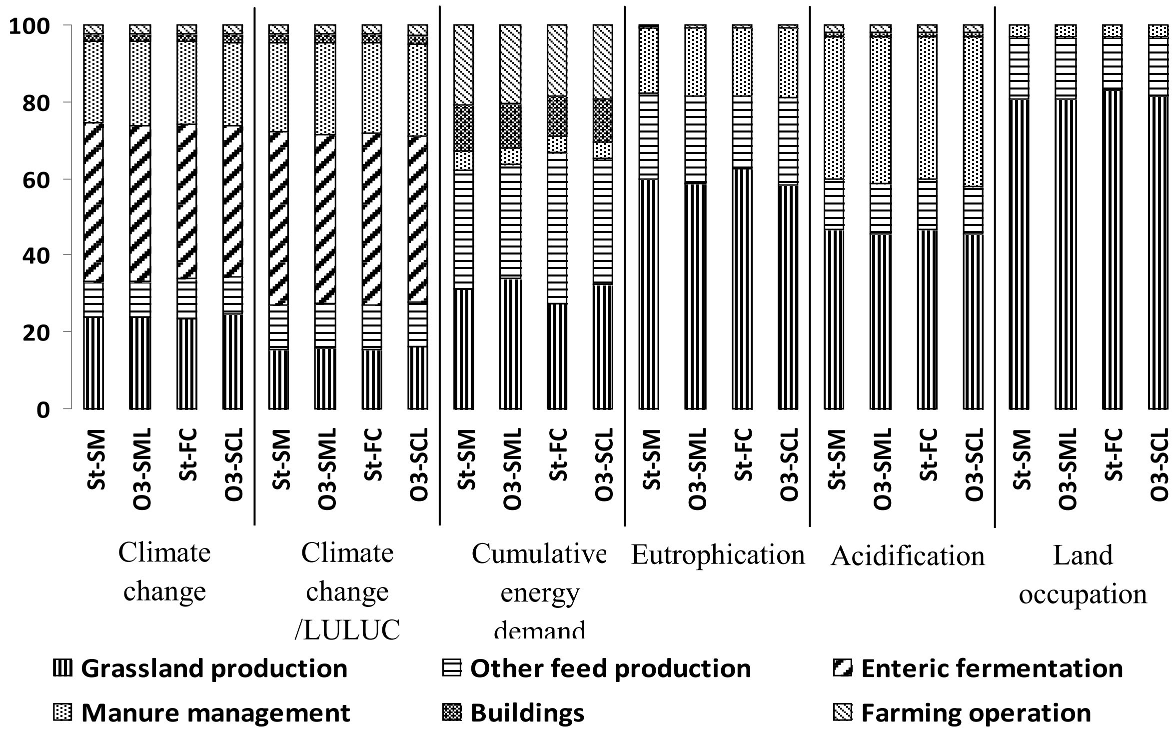

Figure 2: Contribution (in %) of main components in environmental impacts of the four beef-production systems

St: Standard suckler cow-calf herd with finishing heifers
O3: Suckler cow-calf herd with finishing heifers enriched in omega-3 FAs through pasture and wrapped grass silage
SM: Standard bull-fattening herd using a diet rich in starch based on maize silage
SML: Bull-fattening herd using a diet rich in starch (based on maize silage) supplemented with linseeds
FC: Bull-fattening herd using a fibre-based concentrate diet
SCL: Bull-fattening herd using a starch-based concentrate supplemented with linseeds
LULUC: Land use and land-use change

For all scenarios, the cow-calf herd contributed most to the environmental impacts of the beef production system (Fig. 3). The contribution of the cow-calf herd to the impacts per kg of carcass mass was highest for LO (95%), followed by CC (89%), CC/LULUC (87%), EP (88%), AP (85%) and lowest for CED (78%). In general, environmental impacts to produce one kg of live weight gain in a bull-fattening herd (SM, SML, FC and SCL) were lower (-35% to -89%, according to the impact category) than those in a cow-calf herd (St and O3), except for CED in FC (+55%) and SCL (-6%) (Table 6). Nevertheless, when the environmental impacts of each herd are expressed per ha (Table S5), the impacts of the bull-fattening herd were 2-5 times higher than those of the cow-calf herd, except for CED of FC (14 times). Comparing St and O3, the impacts expressed per kg of live weight gain and per ha were higher for O3. In comparing the four bull-fattening herds, all impacts expressed per ha and CED per kg of live weight gain of FC were highest.

Table 6: Impacts per kg of live weight gain produced of two suckler cow-calf herds with finishing heifers and four bull-fattening herds

		Suckler cow-calf herd with finishing heifers		Bull-fattening herd			
		St	O3	SM	SML	FC	SCL
Climate change	kg CO_2 eq	17.5	18.3	8.6	8.0	9.1	6.3
Climate change/LULUC	kg CO_2 eq	15.7	16.3	9.5	8.8	9.1	6.4
Cumulative energy demand	MJ	37.8	41.0	33.1	32.3	58.5	38.7
Eutrophication	g PO_4^{3-} eq	62	63	32	33	19	33
Acidification	g SO_2 eq	103	109	67	65	63	65
Land occupation	m^2/yr	32.1	33.1	7.8	7.4	3.6	6.3

St: Standard suckler cow-calf herd with finishing heifers

O3: Suckler cow-calf herd with finishing heifers enriched in omega-3 FAs through pasture and wrapped grass silage

SM: Standard bull-fattening herd using a diet rich in starch based on maize silage

SML: Bull-fattening herd using a diet rich in starch (based on maize silage) supplemented with linseeds

FC: Bull-fattening herd using a fibre-based concentrate diet

SCL: Bull-fattening herd using a starch-based concentrate supplemented with linseeds

LULUC: Land use and land-use change

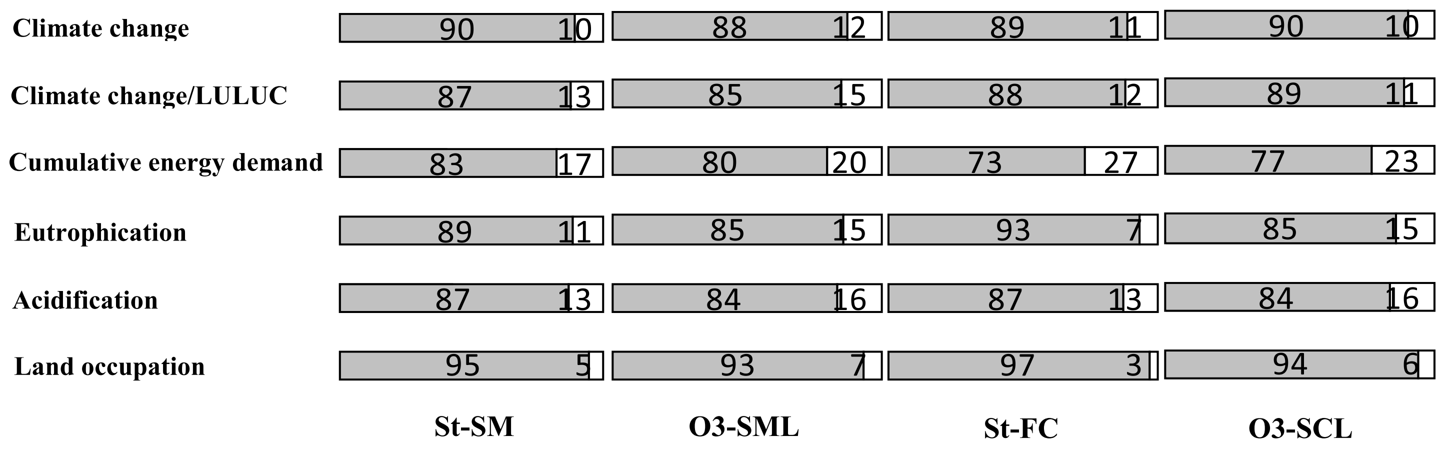

Figure 3: Contribution (%) of suckler cow-calf herd with finishing heifers (grey boxes) and of bull-fattening herd (white boxes) to the environmental impacts of the four beef-production systems

St: Standard suckler cow-calf herd with finishing heifers

O3: Suckler cow-calf herd with finishing heifers enriched in omega-3 FAs through pasture and wrapped grass silage

SM: Standard bull-fattening herd using a diet rich in starch based on maize silage

SML: Bull-fattening herd using a diet rich in starch (based on maize silage) supplemented with linseeds

FC: Bull-fattening herd using a fibre-based concentrate diet

SCL: Bull-fattening herd using a starch-based concentrate supplemented with linseeds

LULUC: Land use and land-use change

3.2. Effect of allocation methods on co-product impacts

The systems delivered carcass mass of four types of animals: fattened bulls (50% total carcass mass), finished heifers (21%), finished cull cows (27%) and a breeding bull (2%). The relative impacts of each type of carcass mass in each system varied according to the allocation method used and the impact considered (Table 7 and Table S6). With mass allocation, for all systems studied, impact values for finished cull cows were highest, followed by those for breeding-bull carcass, finished heifers, and fattened bulls, except for CED of fattened bulls in St-FC. With other allocation methods, impact values for breeding-bull carcass were lowest in all systems. Whatever the allocation method used, impact values for fattened-bull carcass were lower than those for finished cull-cow and finished-heifer carcass, except for CED. For finished cull-cow and finished-heifer carcass, protein allocation yielded higher impact values than economic allocation, but for fattened-bull carcass the opposite occurred.

Economic allocation between beef-product income and agro-environmental subsidies resulted in the attribution of approximately 9% of the impacts of the reproduction component of beef-production systems to ecosystem services (Table 2). Impact of activities to maintain ecosystem services, expressed per ha of grassland, are presented in Table S7. The allocation of impacts to ecosystem services reduced impact values per kg of carcass by 6-9% relative to economic allocation without considering ecosystem services.

Table 7: Climate-change impacts and cumulative energy demand per kg of carcass mass of four types of animals delivered from the four production systems according to four allocation methods

	Fattened bull				Finished heifer		Finished cull cow		Breeding bull	
	St-SM	**O3-SML**	**St-FC**	**O3-SCL**	**St**	**O3**	**St**	**O3**	**St**	**O3**
Climate-change (kg CO_2 eq/kg carcass mass)										
Mass allocation	22.4	22.1	22.8	20.6	31.0	31.2	34.8	34.9	31.9	32.2
Protein allocation	25.0	24.7	25.4	23.2	33.6	33.8	29.2	29.3	15.5	15.6
Economic allocation	25.9	25.6	26.3	24.1	31.9	32.1	28.6	28.6	19.6	19.9
Economic allocation with subsidies	24.2	24.0	24.5	22.5	28.8	29.1	26.3	26.4	17.8	18.1
Cumulative energy demand (MJ/kg carcass mass)										
Mass allocation	59.1	62.7	76.5	68.3	63.5	66.9	76.0	79.5	69.3	73.7
Protein allocation	64.8	68.7	82.2	74.3	69.1	72.9	63.9	66.8	33.7	35.8
Economic allocation	66.7	70.8	84.1	76.4	65.5	69.1	62.5	65.2	42.7	45.4
Economic allocation with subsidies	63.0	67.0	80.4	72.5	59.3	62.5	57.5	60.1	38.8	41.4

St: Standard suckler cow-calf herd with finishing heifers

O3: Suckler cow-calf herd with finishing heifers enriched in omega-3 FAs through pasture and wrapped grass silage

SM: Standard bull-fattening herd using a diet rich in starch based on maize silage

SML: Bull-fattening herd using a diet rich in starch (based on maize silage) supplemented with linseeds

FC: Bull-fattening herd using a fibre-based concentrate diet

SCL: Bull-fattening herd using a starch-based concentrate supplemented with linseeds

Beef-production systems: 1) St-SM, 2) O3-SML, 3) St-FC, 4) O3-SCL

4. Discussion

4.1. Comparison with previous studies

Previous LCA studies on cradle-to-farm-gate beef-production systems show a large variability between impacts. Climate-change impact of the whole suckler beef-production system, without consideration of LULUC, reported from studies in Brazil (Cederberg *et al.*, 2009), the European Union (Nguyen *et al.*, 2010), the United Kingdom (Williams *et al.*, 2006) and Canada (Beauchemin *et al.*, 2010) were 28.2, 27.3, 25.3 and 21.7 kg CO_2 eq/kg carcass mass, respectively. Our results (27.0-27.9; Table 5) are within the range obtained by these authors. Expressed per kg of live weight, CC impact varied from 15.3-15.9 kg CO_2 eq (data not shown) in our study, and are within the range obtained by Pelletier *et al.* (2010) in the United States (US; 14.8-19.2 kg CO_2 eq) and by Veysset *et al*. (2010, 2011) and Gac *et al.* (2010b) in France (14.1-20.2 kg CO_2 eq). For other impacts, our results per kg of carcass mass represented 38-60% for EP, 24-80% for AP, and 112-125% for LO relative to the impact values obtained by Williams *et al.* (2006) and Nguyen *et al.* (2010). Per ha of land occupation, our figures represented 88-92% for CC, 31-55% for EP, and 20-72% for AP relative to the impact values converted from Williams *et al.* (2006) and Nguyen *et al.* (2010). Differences between the present study and literature data can be partly explained by differences between production system characteristics. Our cow-calf herds are extensive production systems in which nearly 80% of the surface was permanent grassland. In our systems, cows are 3-years-old at calving and provide an average of 4.3 calvings per lifetime; the more productive US or Canadian systems provide 6.7 and 6.5 calvings/cow, respectively. Another point is that in the system we studied, weaned female calves not used to replace cows are also reared as heifers to replace cows until the age of 29 months and then they are fattened on pasture until 33 months. Only weaned male calves are intensively fattened to produce bulls. In this study, the results for CC/LULUC are based on data for C sequestration in French agricultural soils; they are far below recent data on grassland C sequestration reported by Soussana *et al*. (2010) for certain European conditions and may underestimate the extent of net C storage in soils. A minor reduction in CC impact (9%) was obtained in this study regardless of the functional unit used. However, Pelletier *et al.* (2010) estimated a decrease in CC impact of 11% to 43% by considering C sequestration in improved pastures (120 kg C/ha/year) and unmanaged pastures (400 kg C/ha/year) under US conditions, but C loss from arable soils converted from pastures was not included. Higher compensation of CC impact (13-21%) was obtained by Veysset *et al.* (2011) because C sequestered in permanent grassland was higher (350 kg C/ha), and C release was considered only for the proportion of cropland converted each year from temporary grassland.

For our systems, the relative contribution of the cow-calf herds to overall impacts was higher than for those reported by Pelletier *et al*. (2010). This is partially due to the higher replacement rate of cows in our systems and to the bull-fattening herd, which concerned only weaned male calves. These results suggest that research emphasis should be put on the cow-calf herds to reduce the environmental impacts of this beef-production system. When the cow-calf herds and the bull-fattening herd are considered separately, the former uses much more land to produce one kg of live weight gain than the latter. However, these cow-calf herds were located on extensive grasslands in the Massif Central region with a low potential for annual crop production. Beef-cattle farming in this region, principally based on permanent grassland, plays an important role in sustaining the rural population and an attractive countryside. This is demonstrated by the low environmental impacts per ha of land for the cow-calf herds, which represented 19-55% of those for the bull-fattening herd, except for CED of St *vs.* FC (7%). Our values for CC per kg of live weight gain for the bull-fattening herd were higher than those reported for a feedlot finishing phase by Phetteplace *et al*. (2001) in the United States and Doreau *et al*. (2011b) in France but lower than those of Ogino *et al*. (2004) in Japanese conditions.

4.2. Effect of omega-3 FA enrichment in the diet and of the proportion of concentrate on environmental impacts

Both per kg of carcass mass and per ha of land occupation, minor differences between the four systems were observed for CC, EP and AP. This can be explained by the high contribution of the cow-calf herd (Fig. 2) on the environmental impacts of these systems and the minor differences between St and O3 (Tables 6 and S5). The production strategy (indoor finishing of cull cows and outdoor finishing of heifers not used for replacement) and the technical characteristics (grassland yield per ha, livestock units per ha of grassland, annual calving rate and replacement rate) were similar for these two cow-calf herds. The differences in the calving period (February or January), the age at which the male calf was sent to the fattening system (11 or 9 months) and the use of forage in winter (only hay or wrapped grass silage and hay) did not greatly differentiate the environmental impacts of St and O3 herds. Apart from replacing hay with wrapped grass silage, there is no other simple and economically viable means to increase omega-3 FAs, as in the suckler cow-calf herds with finishing heifers only a small amount of concentrate is fed to each animal. Among forages, differences in omega-3 FA content are mainly related to the mode of conservation and the age at cutting, and depend to a lesser extent on forage species (Van Ranst et al., 2009).

Differences between systems are larger when the bull-fattening herd is considered alone. The use of rations with 87% concentrate for animals in FC and SCL herds increased CED both per kg of live weight gain and per ha, due to feed-ingredient production and feed processing, compared to the use of rations based on maize silage for animals in SM and SML herds. In the bull-fattening herd, CC was lower with a concentrate diet based on starch (SCL) than with a forage diet based on maize silage (SML) via the strong reduction of enteric methane related to a high proportion of concentrate and a higher-than-average daily gain for bulls. It is known that an increase in proportion of concentrate in the diet decreases enteric methane emissions from ruminants (Martin *et al.*, 2010). Doreau *et al.* (2011b) reported that a strong decrease in enteric methane emissions of fattening bulls fed with an 86% concentrate diet based on maize grain induced a reduction of CC during the fattening phase compared to using a forage diet based on maize silage. However, a reduction of enteric methane produced by bulls fed with a concentrate diet based on fibre (FC) compared to bulls fed with a diet based on maize silage was countered by higher emissions of nitrous oxide and carbon dioxide from dehydration of beet pulp and lucerne. A minor reduction of CC, CED, AP and LO expressed per kg of live weight gain was obtained in SML compared to SM. Feeding a starch concentrate supplemented with extruded linseed (SCL) strongly reduced CC compared to that obtained in FC via a high reduction of enteric methane and a higher average daily gain for bulls in SCL and a higher carbon dioxide emission in FC due to dehydration of beet pulp and lucerne. The SCL feeding strategy had higher EP impact per kg of live weight gain than the FC strategy, due to low nitrate emissions from the production of a fibre-rich concentrate compared to that of a concentrate rich in starch and lipids and a higher yield of lucerne and beet pulp compared to cereals. The high increase of CED in FC resulted from the energy required for lucerne and beet pulp dehydration to produce the fibre-rich concentrate. The impacts of the FC diet may have resulted more from current industrial processes of feedstuffs than from their chemical composition. It should be noted that the fibre-rich concentrate contained 75% co-products (wheat bran, dehydrated beet pulp, wheat middlings, etc.) which can be digested by ruminants and thus avoid feed competition with other livestock and humans.

4.3. Effect of allocation methods on co-product impacts

The choice of allocation methodology for handling the co-products has a decisive effect on LCA results (Cederberg and Stadig, 2003) and is still under debate. Beef-production systems produce four types of animals (fattened bulls, finished heifers, finished cull cows and a breeding bull) which differ not only in production methods but also in economic value and protein content of live weight mass. The question raised was how to determine the environmental impacts of each type of animal in each system. To our knowledge, no published LCA study has yet examined the environmental impacts of different types of animals produced in a beef-production system. According to the ISO recommendation, allocation should be avoided whenever possible by dividing the main process into sub-processes or by expanding the production system to include additional functions related to the co-products (ISO, 2006). Where allocation cannot be avoided, the allocation should be performed by determining physical causal relationships (JRC, 2010) or the market value of the co-products. For dairy-production systems, biological and economic allocation have often been used to allocate impacts of milk and meat products than mass allocation (Yan et al., 2011) and protein allocation, although ISO standards prefer mass and protein allocation to economic allocation. In fact, allocation based on biological rules reflects a physical causal relationship and is recommended first among other physical causalities such as mass and protein. Protein allocation allows comparison of animal products through protein content (de Vries and de Boer, 2009) and reflects that a main function of the beef-production sector is to provide humans with edible protein. In LCA studies, economic allocation is the most common method (de Vries and de Boer, 2009) because products are manufactured corresponding to a demand reflected in their market value (Jolliet *et al.*, 2010).

We therefore analysed the effects of mass, protein and economic allocation on the impacts of four types of animals produced in each system. The allocation approach strongly affected the impacts per kg of carcass mass of breeding bull and, to a much lesser extent, of fattened bulls, finished heifers and finished cull cows. This is because the live weight mass of a breeding bull has lower protein content and economic value than that of the other animal types. The difference in impacts was lowest between protein and economic allocation for fattened bulls, finished cull cows and the breeding bull, and was lowest between mass and economic allocation for finished heifers. Economic allocation could thus be considered a reference allocation method in beef systems.

The process of CAP reforms has reoriented the development of agriculture in Europe towards the principles of rural development and agricultural multifunctionality (Daniel and Perraud, 2009). The "Second Pillar" of the CAP focuses on agro-environmental subventions. These subsidies are intended for landscape management, nature conservation, environmental protection, biodiversity and rural development and concretely reflect social demand toward maintaining grassland with a low stocking rate. LCA has been criticised for considering only "negative" impacts and excluding the positive impacts of agriculture (e.g. Bockstaller *et al.*, 2010). We do believe that this multifunctionality of agriculture, including the provision of ecosystem services, can be included simply by considering such services as co-products. We therefore allocated the impacts of the systems to both their production function (expressed in animal products) and the provision of environmental services (expressed in grassland area). This method resulted in attribution of 9% of the environmental impacts of the reproduction component of beef-production systems to the activities for maintaining ecosystem services. Frequent modifications of CAP reforms result in the adaptation of farming practices to maximise the subsidy (Bélard and Liénard, 2001). Clearly, a modification in agro-environmental subsidies for grassland reflects a modification in social demand regarding the contribution of grasslands on public goods such as biodiversity and landscapes. The allocation of impacts to animal products and to the activities for maintaining ecosystem services will be modified according to the policy adopted. This approach is an initial attempt to consider the ecosystem services provided by farming systems as co-products when estimating the environmental impacts of animal production. A comparable approach has been suggested for Spanish sheep farming systems (Ripoll-Bosch et al., 2011).

5. Conclusions

Our cradle-to-farm-gate study shows that most environmental impacts of beef-production systems emanate from the suckler cow-calf herd with finishing heifers. As a result of the considerable contribution of this herd to the entire system's impacts and the small differences between the standard and omega-3 FA-enriched herds, the environmental impacts of the four investigated systems did not clearly differ, even though those of the bull-fattening herds varied widely. Including effect of land use and land-use change induced a reduction of 9% of climate-change impacts for the entire production system. Use of linseed for the bull-fattening herd did not influence the systems' environmental impacts. This study further revealed that more research for mitigation of the environmental impacts of beef production should focus on the suckler cow-calf herd with finishing heifers.

The allocation approach strongly affected the impacts per kg of carcass mass of a breeding bull and finished cull cows and, to a much lesser extent, those of fattened bulls and finished heifers. The application of economic allocation considering agro-environmental subsidies has shown that the environmental services of farming systems can be considered in LCA studies, which thus can include the positive impacts of farming systems, such as landscape management and biodiversity conservation. This concept needs to be discussed and developed to highlight and preserve the environmentally friendly aspects of farming systems.

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Supplementary materials

Table S1: Description of animal categories, duration in pasture and in housing, and annual ration plan for the Standard (St) and enriched omega-3 FA (O3) suckler cow-calf herds with finishing heifers

Type and number of animal		Duration on pasture (d)		Duration in housing (d)		Annual ration per animal category (t dry matter)											
						Grazed grass		Hay		Wrapped grass silage		Barley grain		Wheat grain		Mix meal[3]	
		St	O3	St	O3	St	O3	St	O3	St	O3	St	O3	St	O3	St	O3
Cows	70	228	228	137	137	199	191	103	53.5	-	49.2	11.9	14.0	12.3	14.2	0.3	0.2
Heifers (>24 mo)	30	76	46	76	106	8.8	4.2	16.1	10.3	-	10.3	1.1	1.5	1.3	1.5	0.2	-
Heifers (12-24 mo)[1]	31	228	228	137	137	43.6	45.7	21.2	13.0	-	12.8	3.5	2.1	3.7	2.1	0.3	-
Heifers (<12 mo)	31	31	61	61	31	3.8	7.6	9.8	2.1	-	2.2	1.6	1.9	1.8	1.9	-	-
Breeding bulls (>24 mo)	3	228	228	137	137	6.2	5.8	6.7	3.4	-	3.8	0.4	0.4	0.6	0.4	0.04	-
Breeding bull (<24 mo)	1	228	228	137	137	2.2	2.6	1.4	0.5	-	0.5	0.4	0.3	0.4	0.3	0.04	-
Calves[2]	62	198	168	76	106	-	-	-	-	-	-	4.5	4.2	10.2	9.3	4.5	4.2
Male pre-finishers	30	-	-	61	-	-	-	3.8	-	-	-	4.2	-	4.2	-	1.3	-
Cull cows	16	-	-	102	102	-	-	13.2	0	-	13.0	5.4	5.0	5.4	5.1	-	-
Growing heifers	14	122	122	0	0	12.8	13.7	-	-	-	-	1.7	1.3	1.6	1.3	-	-

[1]one heifer died

[2]one weaned male calf replaced the breeding bull

[3]Mix meal composition: 30% soybean meal, 40% rapeseed meal and 30% sunflower meal

Table S2: Composition (in %) of fibre-based concentrate (90.2% dry matter) and starch-lipid-based concentrate (88.5% dry matter)

	Fibre-based concentrate	Starch-lipid-based concentrate
Wheat	-	8.6
Barley	2.5	9.7
Maize		28.0
Dehydrated beet pulp	21.5	6.0
Dehydrated lucerne	22.5	-
Wheat bran	28.0	-
Wheat middlings	12.5	3.0
Soybean meal	-	2.0
Rapeseed meal	3.5	21.4
Croquelin®	-	12.0
Other raw materials	7.3	8.0
Minerals	2.2	1.3

Croquelin® composition: 50% extruded linseed, 30% wheat bran and 20% sunflower meal

Table S3: Main inputs used, dry matter yield and nitrate-N emitted (kg/ha, except for irrigation water in m^3/ha) for pastures and the major feed crops[1]

Pasture or crop type	**N mineral**	**N manure**	**P_2O_5** (triple superphosphate)	**K_2O** (potassium chloride)	**CaO**	**Seed**	**Pesticide** (active ingredient)	**Diesel**	**Agricultural machinery**	**Irrigation water**	**Plastic for silage**	**Yield** (dry matter)[2]	**Nitrate-N emitted**
Permanent grassland for St	28	27	19	39	167	0	0	30	5	0	0	5640	20
Temporary grassland for St	33	0	28	58	167	6	0	51	13	0	0	8280	20
Permanent grassland for O3	28	27	19	39	167	0	0	29	5	0	7	5640	20
Temporary grassland for O3	33	0	28	58	167	6	0	50	14	0	12	8280	20
Wheat	171	7	37	24	167	140	2.6	99	23	0	0	5650	40
Barley	129	6	37	26	167	125	2.6	100	24	0	0	5550	40
Silage maize	57	138	31	29	167	20	1.0	91	22	354	16	11000	40
Soybean from Brazil	6	1	80	80	518	53	1.7	76	18	0	0	2708	18
Linseed	70	0	45	25	0	46	0.8	90	22	0	0	1800	40
Sugar beet	103	32	68	146	167	2	3.4	97	22	99	0	18070	40
Lucerne	0	0	119	256	333	8	0.9	73	19	0	0	13970	15

St: Standard suckler cow-calf herd with finishing heifers

O3: Suckler cow-calf herd with finishing heifers enriched in omega-3 FAs through pasture and wrapped silage

[1]Data for grassland and all crops concern a one-year period, except for soybean, where data are for a six-month period

[2]Yield of grassland corresponds to the yield obtained when all grass is machine harvested. 23% and 75% of the yield of permanent and temporary grassland, respectively, was machine harvested as conserved forage (hay and/or wrapped grass silage). Losses during conservation for both hay and wrapped grass silage were assumed to be 6% of the initial DM. Grass not harvested as conserved forage was available for intake by animals during grazing. For several reasons (selective grazing, trampling of grass, unfavourable weather conditions) a part of the grass grown is not ingested, this “loss” corresponded to 31.5% of grass dry matter available for grazing.

Table S4: Environmental impacts due to the production of 1 t dry matter of forages[1] and other feed ingredients

	Climate change	Climate change/LULUC[2]	Cumulative energy demand	Eutrophication	Acidification	Land occupation
	kg CO_2 eq	kg CO_2 eq	MJ	kg PO_4^{3-}eq	kg SO_2 eq	m^2year
Hay from St permanent grassland	365	227	1531	3.0	4.3	1890
Grazed grass from St permanent grassland	433	243	1017	4.0	5.5	2590
Hay from St temporary grassland	198	151	1098	1.7	1.5	1290
Grazed grass from St temporary grassland	231	167	863	2.3	1.9	1770
Hay from O3 permanent grassland	367	229	1509	3.0	4.3	1890
Wrapped silage from O3 permanent grassland	405	267	2771	3.0	4.5	1890
Grazed grass from O3 permanent grassland	438	248	1017	4.0	5.6	2590
Hay from O3 temporary grassland	199	152	1085	1.7	1.5	1290
Wrapped silage from O3 temporary grassland	214	167	1575	1.7	1.6	1290
Grazed grass from O3 temporary grassland	234	170	862	2.3	1.9	1770
Wheat	551	551	3507	4.5	4.9	1630
Barley	475	475	3208	4.2	4.0	1660
Maize silage	279	279	1644	2.6	2.6	920
Starch-lipid-based concentrate[3]	587	606	6344	4.9	4.3	1280
Fibre-based concentrate[3]	686	686	9454	2.4	4.0	1090
Mix meals[4]	566	851	7416	5.0	4.6	1670
Dehydrated Lucerne	961	961	14660	1.2	4.5	740
Dehydrated beet pulp	902	902	14430	0.6	4.7	120
Croquelin®[5]	686	686	6461	9.0	5.3	1770

St: Standard suckler cow-calf herd with finishing heifers
O3: Suckler cow-calf herd with finishing heifers enriched in omega-3 FAs through pasture and wrapped silage
SCL: Bull-fattening herd using a starch-based concentrate supplemented with linseed
FC: Bull-fattening herd using a fibre-based concentrate diet
[1]Impacts correspond to ingested forages and grazed grass from grassland
[2]LULUC: Land use and land use change
[3]See composition in Table S2
[4]Mix meal composition: 30% soybean meal, 40% rapeseed meal and 30% sunflower meal
[5]Croquelin® composition: 50% extruded linseed, 30% wheat bran and 20% sunflower meal

Table S5: Impacts per ha of two suckler cow-calf herds with finishing heifers and four bull-fattening herds

		Suckler cow-calf herd with finishing heifers		Bull-fattening herd			
		St	**O3**	**SM**	**SML**	**FC**	**SCL**
Climate change	t CO_2 eq	5.5	5.5	11.1	10.8	24.9	10.1
Climate change/LULUC	t CO_2 eq	4.9	4.9	12.2	11.8	24.9	10.2
Cumulative energy demand	GJ	11.8	12.4	42.6	43.4	160.6	61.9
Eutrophication	kg PO_4^{3-} eq	19	19	42	45	53	53
Acidification	kg SO_2 eq	32	33	86	87	172	104

St: Standard suckler cow-calf herd with finishing heifers

O3: Suckler cow-calf herd with finishing heifers enriched in omega-3 FAs through pasture and wrapped grass silage

SM: Standard bull-fattening herd using a diet rich in starch based on maize silage

SML: Bull-fattening herd using a diet rich in starch (based on maize silage) supplemented with linseeds

FC: Bull-fattening herd using a fibre-based concentrate diet

SCL: Bull-fattening herd using a starch-based concentrate supplemented with linseeds

LULUC: Land use and land use change

Table S6: Eutrophication, acidification and land occupation per kg of carcass mass of four animal types delivered from the four production systems according to four allocation methods

	Fattened bull				**Finished heifer**		**Finished cull cow**		**Breeding bull**	
	St-SM	**O3-SML**	**St-FC**	**O3-SCL**	**St**	**O3**	**St**	**O3**	**St**	**O3**
Eutrophication (g PO_4^{3-} eq)										
Mass allocation	81	82	72	82	110	108	120	119	115	114
Protein allocation	91	92	82	92	119	118	100	99	56	55
Economic allocation	94	95	85	95	113	112	98	97	71	70
Economic allocation with subsidies	88	89	79	89	103	101	90	89	64	64
Acidification (g SO_2 eq)										
Mass allocation	144	147	141	147	182	186	204	207	189	194
Protein allocation	160	163	157	163	198	202	171	174	92	94
Economic allocation	165	169	162	169	188	191	167	170	117	119
Economic allocation with subsidies	155	158	152	159	170	173	153	156	106	109
Land occupation (m^2year)										
Mass allocation	35.6	34.7	32.8	33.7	59.2	58.5	61.4	59.9	61.1	60.1
Protein allocation	40.6	39.6	37.8	38.6	64.2	63.4	50.7	49.5	29.7	29.2
Economic allocation	42.3	41.3	39.5	40.3	61.0	60.2	49.4	48.3	37.7	37.1
Economic allocation with subsidies	39.1	38.2	36.3	37.1	55.2	54.5	45.2	44.2	34.2	33.7

St: Standard suckler cow-calf herd with finishing heifers

O3: Suckler cow-calf herd with finishing heifers enriched in omega-3 FAs through pasture and wrapped silage

SM: Standard bull-fattening herd using a diet rich in starch based on maize silage

SML: Bull-fattening herd using a diet rich in starch (based on maize silage) supplemented with linseeds

FC: Bull-fattening herd using a fibre-based concentrate diet

SCL: Bull-fattening herd using a starch-based concentrate supplemented with linseed

Table S7: Impacts of beef meat product (per kg of carcass mass) delivered from the four beef-production systems and of activities for maintaining ecosystem services (per ha of grassland) from two suckler cow-calf herds with finishing heifers using economic allocation with agro-environmental subsidies

	Beef				**Activities for maintaining ecosystem services**	
	St-SM	**O3-SML**	**St-FC**	**O3-SCL**	**St**	**O3**
	per kg carcass mass				per ha of grassland	
Climate change (kg CO_2 eq)	25.6	25.6	25.8	24.8	557	563
Climate change/LULUC (kg CO_2 eq)	23.6	23.6	23.4	22.5	495	501
Cumulative energy demand (MJ)	60.2	63.6	68.8	66.3	1192	1261
Eutrophication (g PO_4^{3-}eq)	91	91	86	91	1984	1966
Acidification (g SO_2 eq)	157	160	155	160	3293	3367
Land occupation (m^2year)	44.0	43.2	42.6	42.7	1052	1039

St: Standard suckler cow-calf herd with finishing heifers

O3: Suckler cow-calf herd with finishing heifers enriched in omega-3 FAs through pasture and wrapped silage

SM: Standard bull-fattening herd using a diet rich in starch based on maize silage

SML: Bull-fattening herd using a diet rich in starch (based on maize silage) supplemented with linseeds

FC: Bull-fattening herd using a fibre-based concentrate diet

SCL: Bull-fattening herd using a starch-based concentrate supplemented with linseeds

LULUC: Land use and land use change

Chapter 3

Effect of farming practices for greenhouse gas mitigation and subsequent alternative land-use on environmental impacts of beef-cattle production systems

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Effect of farming practices for greenhouse gas mitigation and subsequent alternative land-use on environmental impacts of beef-cattle production systems

Short title: Practices and land use to reduce beef farm impacts

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Abstract

This study evaluated effects of farming-practice scenarios aiming to reduce greenhouse gas (GHG) emissions and subsequent alternative land-use on environmental impacts of a beef-cattle production system using the life cycle assessment approach. The baseline scenario includes a standard cow-calf herd with finishing heifers based on grazing, and a standard bull-fattening herd using a diet mainly based on maize silage, corresponding to current farm characteristics and management by beef farmers in France. Alternative scenarios were developed with changes in farming practices. Some scenarios modified grassland management (S1: decreasing mineral N fertiliser on permanent grassland; S2: decreasing grass losses during grazing) or herd management (S3: underfeeding of heifers in winter; S4: fattening female calves instead of being reared at a moderate growth rate; S5: increasing longevity of cows from 7 to 9 years; S6: advancing first calving age from 3 to 2 years). Other scenarios replaced protein sources (S7: partially replacing a protein supplement by lucerne hay for the cow-calf herd, S8: replacing soybean meal with rapeseed meal for the fattening herd) or increased omega-3 fatty-acid content using extruded linseed (S9). The combination of compatible scenarios S1, S2, S5, S6 and S8 was also studied (S10). The impacts, such as climate change (CC, not including CO_2 emissions/sequestration of land use and land-use change, LULUC), CC/LULUC (including CO_2 emissions of LULUC), cumulative energy demand (CED), eutrophication (EP), acidification and land occupation (LO) were expressed per kg of carcass mass and per ha of land occupied. Compared to the baseline, the most promising practice to reduce impacts per kg carcass mass was S10 (all reduced by 13-28%), followed by S6 (by 8-10%). For other scenarios, impact reduction did not exceed 5%, except for EP (up to 11%) and LO (up to 10%). Effects of changes in farming practices (the scenarios) on environmental impacts varied according to impact category and functional unit. For some scenarios (S2, S4, S6 and S10), permanent grassland area and LO per kg of carcass decreased by 12-23% and 9-19%, respectively. If the “excess” permanent grassland were converted to fast-growing conifer forest to sequester carbon in tree and soil biomass, CC/LULUC per kg of carcass could be reduced by 20, 25, 27 and 48% for scenarios S2, S4, S6 and S10, respectively. These results illustrate the potential of farming practices and forest as an alternative land-use to contribute to short and mid-term GHG mitigation of beef-cattle production systems.

Keywords: beef cattle, farming-practices, alternative land-use, environmental impacts, life cycle assessment

Implications

To decrease environmental impacts of beef cattle production systems, different strategies of forage or herd management and of alternative feeding can be proposed. Each of them decreases one or several impacts to a small extent. Strategies have more influence on the whole system when applied to the cow-calf herd than to fattened animals. The most promising strategy is calving at 2 years instead of 3 years. A significant decrease in impacts can be achieved by simultaneously applying several compatible strategies. Some strategies produce the same quantity of meat on less land, and if an increase in meat or crop production is not desired, this "excess" land could be converted to forest to stock carbon, thus decreasing the net greenhouse-gas emissions of the system.

1. Introduction

Livestock production worldwide, in particular ruminant production (Steinfeld et al., 2006; Gill et al., 2010), is responsible for significant greenhouse gas (GHG) emissions. Numerous GHG mitigation strategies for ruminant production have focused on a single GHG such as enteric methane (CH_4) or nitrous oxide (N_2O) (Martin et al., 2010; Eckard et al., 2010). Measures to enhance carbon (C) sequestration in the soil have also been identified (Dawson and Smith, 2007) as a mitigation strategy. However, it is critical to ensure that there is a net reduction in GHG emissions of the whole production system when such measures are implemented (Beauchemin et al., 2011), i.e. that a reduction in on-farm GHG emissions is not compensated by an increase in off-farm GHG emissions due to imported feed. Therefore, these measures need to be assessed at the scale of the entire production system. Besides GHG emissions, other environmental impacts such as energy use, eutrophication and land-use impacts may be of major importance depending on the local or regional context (Steinfeld et al., 2010).

The present study analysed environmental impacts of farming practices meant to reduce GHG emissions of beef-cattle production systems using the life cycle assessment (LCA) approach. The baseline beef production scenario, described by Nguyen et al. (2012a) (chapter 2), reflected current farm characteristics and management practices by farmers of Charolais beef cattle in France. Alternative land-use was assessed by assuming that any permanent grassland becoming available due to more efficient farming practices was converted to even-aged forest.

Nine scenarios were assessed, as well as an aggregated one representing the sum of scenarios considered compatible.

2. Materials and methods

2.1. System boundaries

Life cycle assessments of beef-cattle production systems were conducted from cradle to farm-gate for a one-year period, i.e. including the production and delivery of inputs used for grassland and cereals produced on-farm and for feed produced off-farm, herd management and associated upstream processes, emissions from the animals and manure storage. Environmental impacts from the application of manure for cereals and pasture was included, as were those from buildings. Veterinary medicines were excluded due to lack of data. The impacts, i.e. climate change (CC, excluding and including the effects of land use and land-use change (LULUC)), cumulative energy demand (CED), eutrophication (EP), acidification (AC) and land occupation (LO), of different farming-practice scenarios were compared. The functional units (FUs) considered were “1 kg of carcass mass at the farm exit gate” and “1 ha of on-farm and off-farm land occupied”. If farming practices reduced permanent grassland occupation per kg of carcass mass produced, this released land was converted to fast-growing even-aged conifer forest as an alternative land-use to increase the amount of C sequestered by the farm system. Planting and main management stages for Corsican pine (*Pinus nigra* subsp. *laricio*) were assumed and amortized over 64 years, the mean rotation period for plantations of this species (Vallet et al., 2009).

2.2. Description of baseline of beef-production system

The baseline beef-production system (corresponding to system St-MS described in Nguyen et al. (2012a)) comprised a cow-calf herd and a bull-fattening herd. The cow-calf herd included 70 cows that produced 62 weaned calves each year. These cows had their first calving at 3 years, and each provided a mean of 4.4 calves over their lifetimes. All weaned female calves were reared as heifers (with 3% mortality) used as replacement cows until the age of 27 months. Of the 30 heifers thus produced, 14 were not selected for replacement and were fattened in pasture complemented with cereals and slaughtered at 33 months. Cull cows were finished for 100 days before being sent to the slaughterhouse. One male calf was selected to replace the breeding bull, and the rest were sent to the bull-fattening herd at 11 months and slaughtered at 18 months.

The cow-calf herd ration was based mainly on grassland with a mean of 1.2 livestock units (LU) per ha of grassland (temporary + permanent) and 7.5 months of grazing. The grassland area consisted of 88% permanent and 12% temporary pastures. One LU is defined as an animal that consumes 5 t dry matter (DM)/year (Nguyen et al., 2012a). We assumed that permanent grassland did not require tilling and sowing operations. Apart from manure excreted on pasture during grazing, permanent grassland was fertilised with mineral and organic N fertilisers (contributing 28 and 27 kg/ha of N, respectively). Permanent grassland had a potential yield of 5.6 t DM/ha/year, 23% of which was harvested as conserved forage (hay and/or wrapped grass-silage). Temporary grassland, a combination of grasses and clover, had a higher potential yield (8.3 t DM/ha/year, 75% harvested as conserved forage) and was renewed every 5 years by tillage and seeding. Mineral N fertiliser for temporary grassland was applied at 33 kg/ha. Grass not harvested as conserved forage was available for ingestion by animals during grazing. For several reasons (selective grazing, trampling of grass, unfavourable weather conditions), some of the grass grown is not ingested; this "loss" corresponded to 31.5% of grass DM available for grazing, as calculated from the difference between grassland potential yield and actual feed intake by the herd. Losses during conservation of both hay and wrapped grass-silage were assumed to be 6% of the initial DM of conserved forages. During the indoor winter-feeding period, the herd was fed hay and concentrates (mainly based on cereals produced on-farm and imported protein supplement containing 30% soybean meal, 40% rapeseed meal and 30% sunflower meal).

Male calves in the baseline bull-fattening herd were fed a high-forage diet composed of 58% maize silage, 24% wheat, 15% soybean meal, 2% hay, and 1% minerals (DM basis), resulting in an average daily live weight gain (ADG) of 1.40 kg. All rations were formulated to satisfy beef-cattle nutrient requirements according to animal characteristics and feed-composition values, based on recommendations of INRA beef researchers and data tables (INRA, 2007). The carcass yields of fattened bulls, the breeding bull, finished heifers and finished cull cows were 59%, 57%, 56% and 54%, respectively. Methods used to produce feed ingredients were described in Nguyen et al. (2012a) and were summarised in supporting information Table S1.

2.3. Scenarios with alternative farming practices

Scenarios with alternative farming practices (denoted S1 to S10) were designed to reduce GHG emissions of the beef-cattle production system. These practices are already applied by some farmers or can be applied without adverse effect on animal performances, based on experimental results. The use of these practices, both individually and simultaneously was studied. Alternative rations were formulated according to INRA (2007) to meet animal requirements, except in scenario S3 (underfeeding). When farming practice affected total feed requirements, the land area needed was adjusted to produce feed. Feed ingredients were produced by the same practices used in the baseline scenario.

Grassland management

Scenario S1. Mineral N fertiliser decreased. This scenario assessed effects of decreasing mineral N fertiliser from 28.0 to 18.5 kg/ha of permanent grassland. The yield of permanent grassland was assumed not to be affected because baseline mineral N fertiliser application levels exceed the optimum level required for grass growth (J. Devun, Institut de l'Elevage, pers. comm.). Estimated nitrate losses through leaching were reduced from 20 to 14 kg N/ha. As grassland yield was not affected, this reduction did not change land use or reproduction or growth performances of grazing animals.

Scenario S2. Grass losses on pasture decreased. This scenario evaluated effects of decreasing grass losses (i.e. grass that is not ingested by the cows) on pasture from 31.5 to 16.5% (J. Devun, Institut de l'Elevage, pers. comm.). This reduction can be obtained by better management of grassland, i.e. turn out to pasture as soon as possible, rotational grazing, adjust animal density for grazing during the dry season (Joannic et al., 2011). As a consequence, the stocking rate was increased from 1.20 to 1.37 LU/ha of grassland area. Estimated nitrate losses were decreased from 20 to 17 kg N/ha. It was assumed that this practice did not affect reproduction and growth performances of grazing animals.

Herd management

Scenario S3. Underfeeding of heifers in winter. This scenario evaluated effects of underfeeding of heifers in winter using exclusively hay, and animal growth was assumed to be compensated during the grazing season. Rations were formulated by INRAtion v.4, and heifer growth was predicted with the Mecsic model (Hoch and Agabriel, 2004). Stocking rate was decreased from 1.20 to 1.15 LU/ha of grassland area.

Scenario S4. Female calves fattened (high growth rate) instead of being reared for replacement (moderate growth rate). This scenario evaluated effects of fattening of female calves from 9 to 19 months instead of rearing them as heifers used for replacement and fattening them on pasture for 4 months until slaughter at 33 months. Fourteen female calves after weaning not selected for replacement were fattened (until 650 kg LW) with a diet based on maize silage (76.5% maize silage, 1.3% hay, 13.6% wheat, 7.0% soybean meal, and 1.6% minerals (DM basis)), resulting in an ADG of 1.15 kg.

Scenario S5. Cow longevity increased. This scenario evaluated effects of increasing longevity of cows from 7 to 9 years to provide a mean of 6.5 calves per lifetime instead of 4.4 calves. As a consequence, the number of culled cows decreased (from 16 to 11 per year), and the number of heifers used for meat production increased (from 14 to 19 per year). This practice is assumed to be achieved by changes in farm management and not to affect calving rate, animal growth or mortality of the herd, according to the experience of farmers that implemented this approach.

Scenario S6. Age at first calving decreased. This scenario evaluated effects of decreasing first calving age from 3 to 2 years simulated based on Farrié et al. (2008). All female calves were reared to reach 467 kg LW (instead of 405 kg) at 15 months for the first breeding. Heifers not used for replacement at 15 months were fattened to slaughter at 23 months (about 670 kg LW) instead of 33 months (at 698 kg LW). Replacement rate was slightly lower (21.4%) than in the baseline (23%) scenario; although these cows produced more calves (mean = 4.7 instead of 4.4) per lifetime, they were culled sooner (at 6 years and 780 kg LW instead of at 7 years and 800 kg LW). According to farmer's experiences, under normal conditions, this practice can be achieved by changes in farm management without affecting calving rate and mortality of the herd.

Feed composition

Scenario S7. Protein supplement partially replaced with lucerne hay. This scenario evaluated the effects of replacing some protein supplement with lucerne hay during the winter. A portion of temporary grassland was used to produce lucerne hay, and the protein supplement for the herd per year was decreased from 6.8 to 2.3 t. Lucerne hay contributed 12.4% of the total hay production. It was assumed that this practice did not affect reproduction and growth performances because total digestible protein intake was unchanged.

Scenario S8. Soybean meal replaced with rapeseed meal. This scenario evaluated effects of using rapeseed meal to replace soybean meal in the bull diet. It was assumed that animal growth was not affected, because nutrient intake per day was maintained by increasing DM intake.

Scenario S9. Lipid content in diets increased by using extruded linseed. Extruded linseed was used to replace a portion of concentrate (cereals and protein supplement) in the cow-calf herd. Lipid content in diets for animals was not to exceed 3% of total DM. As animal requirements were met in both diets, it was assumed that this practice did not affect animal performances during winter. Male calves were sent to the bull-fattening herd after weaning (350 kg LW) and were fed with concentrate-based diet rich in lipids (13% barley straw and 83% concentrate including 46% cereals and 6% extruded linseed) resulting an ADG of 1.71 kg.

Scenario S10. Combination of scenarios S1, S2, S5, S6 and S8. Scenario S10 combines five compatible scenarios whose effects were expected to be additive: decrease in mineral N fertiliser (S1), decrease in grass losses on pasture (S2), increase in cow longevity from 7 to 9 years (S5), decrease in age at first calving from 3 to 2 years (S6) and replacement of soybean meal with rapeseed meal (S8). Details of baseline and farming-practice scenarios of the beef production system are presented in Table 1.

Table 1: Annual characteristics of baseline and farming-practice scenarios* for the beef-cattle production system

Scenarios	Baseline**	S1	S2	S3	S4	S5	S6	S7	S8	S9	S10
Feed intake											
Pastured grass intake, t DM***	276.5	276.5	276.5	286.1	235.0	275.0	241.5	276.5	276.5	271.1	241.3
Hay, t DM	176.6	176.6	176.6	190.3	155.4	172.5	150.3	162.2	175.4	182.5	147.8
Lucerne hay, t DM	-	-	-	-	-	-	-	22.9	-	-	-
Maize silage, t DM	31.7	31.7	31.7	31.7	61.9	31.7	31.7	31.7	34.8	0.0	34.8
Cereals, t DM	91.5	91.5	91.5	76.4	87.6	89.3	92.6	90.9	88.3	80.0	83.9
Imported soybean meal, t DM	11.2	11.2	11.2	11.1	14.0	11.2	11.3	9.9	2.0	2.3	2.1
Imported rapeseed meal, t DM	2.7	2.7	2.7	2.5	2.6	2.7	2.8	0.9	13.3	12.9	13.4
Imported sunflower meal, t DM	2.0	2.0	2.0	1.9	2.0	2.0	2.1	0.7	2.0	2.5	2.1
Imported extruded linseed, t DM	-	-	-	-	-	-	-	-	-	11.5	-
Imported other raw materials, t DM	0.9	0.9	0.9	0.9	1.5	0.9	0.9	0.9	0.9	18.9	0.9
Grassland management											
Grass losses**** on pasture, %	31.5	31.5	16.5	31.5	31.5	31.5	31.5	31.5	31.5	31.5	16.5
Mineral N fertiliser for permanent grassland, kg/ha	28.0	18.5	28.0	28.0	28.0	28.0	28.0	28.0	28.0	28.0	18.5
Herd characteristics											
Replacement rate, %	23.0	23.0	23.0	23.0	23.0	15.5	21.4	23.0	23.0	23.0	13.0
Age of first calving, years	3	3	3	3	3	3	2	3	3	3	2
Number of animals produced x Live weight, kg/animal											
Breeding bull	1 x 990	1 x 990	1 x 990	1 x 990	1 x 990	1 x 990	1 x 990	1 x 990	1 x 990	1 x 990	1 x 990
Finished cull cows	16 x 798	16 x 798	16 x 798	16 x 798	16 x 798	11 x 818	15 x 792	16 x 798	16 x 798	16 x 802	9 x 824
Finished heifers (33 mo)	14 x 695	14 x 695	14 x 695	14 x 695	-	19 x 695	-	14 x 695	14 x 695	14 x 701	-
Finished heifers (23 mo)	-	-	-	-	-	-	15 x 668	-	-	-	21 x 668
Finished heifers (19 mo)	-	-	-	-	14 x 650	-	-	-	-	-	-
Finished bulls (18 mo)	30 x 720	30 x 720	30 x 720	30 x 720	30 x 720	30 x 720	30 x 720	30 x 720	30 x 720	30 x 720	30 x 720
Total carcass mass, t	25.7	25.7	25.7	25.7	25.3	25.6	25.3	25.7	25.7	25.7	25.2

*Scenarios are defined in Material and Methods

**Nguyen et al. (2012a)

*** Dry matter

**** Grass that not ingested by the animals on pasture

Alternative land use: Corsican pine even-aged forest

If an alternative scenario used less land than the baseline scenario to produce the same quantity of meat (carcass mass), we explored an alternative use for this “released” land -- to reduce net GHG emissions of the farm system -- rather than to use it to increase meat or crop production. We assumed that the surplus land area was converted to an even-aged forest of Corsican pine, because it grows well even on poor sites, provides high-quality wood and has been successful in several French regions. We assumed a 64-year rotation, during which the forest sequesters 11.4 t CO_2/ha/yr into the vegetation (Vallet et al., 2009). The main function of the forest within the beef-farm system being C sequestration, we did not include the harvest of the trees (after 64 years), neither concerning inputs required nor the products it would yield. We did, however, include inputs required for planting the forest and managing it during the first 15 years of the establishment phase.

2.4. Emissions estimates, including effect of LULUC on soil C balance

Methods for estimating farm emissions were described in Nguyen et al. (2012a). Briefly, enteric CH_4 emissions were estimated for each class of cattle according to Vermorel et al. (2008) using animals’ net-energy requirements, converted into metabolisable energy intake (MEI) and conversion factors from MEI to CH_4 energy. To include the effect of diets supplemented with lipids rich in omega-3 fatty acids (FAs) on ruminants’ enteric CH_4 production, a 4.8% reduction factor of enteric CH_4 production (g CH_4/kg DM intake) per percentage unit of added lipids was applied (Martin et al., 2010).

Emissions from manure produced by cattle (manure in the cow-calf herd and slurry in the bull-fattening herd) in housing, during storage, deposited during grazing and from manure application on cropland and grassland were estimated according to IPCC (2006) Tier 2 (for CH_4 and N_2O), CORPEN (2006) and Payraudeau et al. (2007) (for ammonia). Nitrate leaching was estimated based on Vertès et al. (2007) for grassland and Basset-Mens et al. (2007) for cropland. Phosphorus emissions (leaching, run-off and erosion) were estimated according to Nemecek and Kägi (2007). A summary of emission factors used for livestock, cropping and grassland production and their sources is presented in Table S2.

C sequestration according to type of grassland was estimated using data from Arrouays et al. (2002) for permanent grassland (i.e. older than 30 years, 0.7 t CO_2/ha/yr) and for temporary grassland (1.8 t CO_2/ha/yr). It was assumed that temporary grassland was maintained for five years and followed by an annual crop for two years; C emissions were estimated at 3.7 t CO_2/ha/yr for this cropland in rotation with temporary grassland (Arrouays et al., 2002). We assumed that other annual-crop area was converted from permanent grassland more than 20 years ago, and agricultural practices for these crops no longer had an effect on soil carbon. The proportion of Brazilian soybean crops grown on land converted the previous year from Brazilian rain forest was estimated at 0.7% (Prudêncio da Silva et al., 2010). To conform better to current practice regarding the effect of land-use change on C emissions due to conversion of Brazilian forest to cropland, we decided to adopt a value of 740 t CO_2/ha, as recommended in PAS 2050 (2008) among others, instead of the value of 120 t CO_2/ha used in the ecoinvent database (Jungbluth et al., 2007).

2.5. Life cycle impact assessment

The impact categories considered were LO (m^2*yr), CC and CC/LULUC (kg CO_2 equivalent (eq.)), CED (MJ), EP (g PO_4^{3-} eq.) and AC (g SO_2 eq.). The indicator value for each impact category was determined by multiplying the aggregated resources used and the aggregated emissions of each individual substance with a characterisation factor for each impact category to which it may potentially contribute. CC, EP, AC and LO were calculated using the CML2 "baseline" and "all categories" 2001 characterisation methods as implemented in the ecoinvent v2.0 database. The CC indicator excludes C sequestration in grassland, that in even-aged forest converted from permanent grassland and C emissions due to conversion of Brazilian forest to cropland, whereas the CC/LULUC includes them. Total CED was calculated according to version 1.05 of the indicator, as implemented in the ecoinvent v2.0 database.

3. Results

3.1. Effects of changes in farming practices on CC, CC/LULUC, CED, EP and AC

Effects of grassland management (S1-S2). Decreasing mineral N fertiliser application on permanent grassland (S1) slightly decreased CC, CC/LULUC and AC (reduction between 1 and 2%) and decreased CED and EP per kg carcass mass by 2.9 and 10.5%, respectively

(Table 2). The reduction of CC and CC/LULUC was due mainly to lower N_2O emissions (Table 3) associated with reduced mineral N fertiliser application. Decreasing grass losses on pasture (S2) did not affect CC/LULUC (reduction <1%), slightly decreased CC and AC, and decreased CED and EP per kg carcass mass by 2.8 and 10.8%, respectively. The reduction of CC and CC/LULUC was due to lower N_2O emission from mineral fertiliser application on grassland, as less grassland was needed to produce one kg of carcass mass. However, the reduction in grassland area induced a decrease in total C sequestration by the beef system.

*Effect of herd management (S3-S6).*Underfeeding heifers in winter (S3) did not affect CC, AC, and slightly decreased CC/LULUC, CED and EP. Fattening female calves instead of rearing them as replacement heifers (S4) slightly decreased CED, and decreased impacts per kg carcass mass by 4.9, 3.5, 4.4 and 3.6% for CC, CC/LULUC, EP and AC, respectively. The reduction of CC and CC/LULUC was related to lower emissions of enteric CH_4 and N_2O emissions from feed production and manure. However, CH_4 emissions from manure increased and C sequestration decreased. Increasing cow longevity (S5) slightly decreased impacts per kg carcass mass. Decreasing calving age (S6) decreased impacts per kg carcass mass about 7.8-8.4%. The reduction of CC and CC/LULUC was related principally to the reduction of enteric CH_4 (by 8%), N_2O emissions from feed production (by 9%) and manure (by 9%) and CO_2 emission from fossil-fuel use (by 8%). However, C sequestration decreased by 15%.

Effect of feed composition (S7-S9). The partial replacement of protein supplement by lucerne hay during the winter (S7) did not affect any impact category per kg carcass mass. The replacement of soybean meal by rapeseed meal in bull diets (S8) did not affect CC, EP and AC, slightly decreased CC/LULUC and reduced CED per kg carcass mass by 3.8%. A reduction of fossil-fuel-based CO_2 emissions and a net increase in C sequestration were partially compensated by an increase in N_2O emission from feed production and CH_4 emission from manure. The use of extruded linseed to increase lipid content in animal diets (S9) slightly decreased AC, decreased CC and CC/LULUC per kg carcass mass by 3.0 and 4.4%, but increased CED and EP by 8.0 and 6.7%, respectively. Emissions of enteric CH_4 and CH_4 from manure decreased by 9 and 8%, respectively. However, CH_4 and CO_2 emission from fossil-fuel use and N_2O emission from manure increased by 31, 7 and 4%, respectively. C sequestration also increased by 12%.

Table 2: Environmental impacts of baseline for standard beef-cattle production and farming-practice scenarios*

Impact	Unit	**Baseline****	**S1**	**S2**	**S3**	**S4**	**S5**	**S6**	**S7**	**S8**	**S9**	**S10**
Per kg carcass mass												
Climate change	kg CO_2 eq.	27.8	27.3	27.2	27.6	26.4	27.4	25.5	27.9	27.8	26.9	24.2
Climate change/ LULUC***	kg CO_2 eq.	25.5	25.0	25.3	25.2	24.6	25.1	23.5	25.5	25.2	24.4	22.2
Cumulative energy demand	MJ eq.	65.0	63.1	63.2	64.1	64.2	64.3	59.8	64.7	62.6	70.2	53.4
Eutrophication	g PO_4^{3-} eq.	98.7	88.3	88.1	97.8	94.4	97.3	90.7	98.1	98.0	105.4	71.0
Acidification	g SO_2 eq.	169.5	167.4	166.4	169.3	163.5	167.3	155.3	170.1	168.5	173.1	147.9
Land occupation	m^2*year	48.2	48.2	43.4	49.0	43.7	47.7	43.4	48.3	48.1	47.4	38.9
Per ha of land occupied												
Climate change	t CO_2 eq.	5.77	5.66	6.28	5.64	6.05	5.75	5.87	5.74	5.78	5.69	6.22
Climate change/ LULUC*	t CO_2 eq.	5.29	5.19	5.83	5.14	5.64	5.28	5.42	5.27	5.24	5.14	5.72
Cumulative energy demand	GJ eq.	13.50	13.11	14.57	13.08	14.70	13.49	13.78	13.48	13.01	14.82	13.73
Eutrophication	kg PO_4^{3-} eq.	20.49	18.34	20.30	19.94	21.61	20.40	20.92	20.38	20.38	22.23	18.25
Acidification	kg SO_2 eq.	35.20	34.76	38.35	34.53	37.44	35.05	35.82	35.25	35.02	36.52	38.02

*Scenarios are defined in Material and Methods
**Nguyen et al. (2012a)
***Land use and land-use change

Table 3: Contribution of CH_4, N_2O and CO_2 to climate change impact (kg CO_2 eq./kg carcass mass) (including LULUC*) of baseline for standard beef-cattle production and farming-practice scenarios**

Emission	**Baseline*****	**S1**	**S2**	**S3**	**S4**	**S5**	**S6**	**S7**	**S8**	**S9**	**S10**
Enteric CH_4	11.49	11.49	11.49	11.42	10.96	11.35	10.58	11.49	11.49	10.46	10.45
CH_4 from manure	0.82	0.82	0.82	0.83	1.10	0.81	0.78	0.82	0.88	0.75	0.84
CH_4 from fossil fuels	0.16	0.16	0.16	0.16	0.16	0.15	0.15	0.16	0.15	0.21	0.13
N_2O from feed	7.30	6.87	6.86	7.30	6.73	7.24	6.62	7.23	7.36	7.07	5.88
N_2O from manure	4.61	4.61	4.61	4.55	4.14	4.47	4.21	4.75	4.61	4.80	4.09
CO_2 from fossil fuels	3.38	3.29	3.28	3.34	3.31	3.34	3.10	3.38	3.27	3.61	2.80
CO_2 including LULUC***	-2.29	-2.29	-1.96	-2.42	-1.82	-2.27	-1.95	-2.33	-2.59	-2.57	-1.98
Others	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.04	0.02

* Land use and land-use change
** Scenarios are defined in Material and Methods
*** Nguyen et al. (2012a)

Scenario S10. Combination of S1, S2, S5, S6 and S8 scenarios. The combination of S1, S2, S5, S6 and S8 decreased CC, CC/LULUC and AC per kg carcass mass by 12.7-12.8%, CED by 17.9% and EP by 28.0%. There was a high reduction (by 9-27%) in emission of all GHGs, except for CH_4 emission from manure (by 5%); however, C sequestration decreased by 14%.

The effects of farming practices S1, S7 and S8 on the environmental impacts per ha of on- and off-farm land occupied were approximately the same as those per kg of carcass mass (Table 2). In contrast, the effects of the other scenarios differed according to FU. Decreasing grass loss on p asture (S2) increased impacts per ha by 8-10%, except for EP (-1%). Underfeeding heifers in winter (S3) decreased impacts per ha by 2-3%. Fattening female calves instead of rearing them as replacement heifers (S4) increased impacts per ha by 5-9%. Increasing cow longevity (S5) did not affect impacts per ha. Decreasing calving age (S6) increased impacts per ha by only 2%. The use of extruded linseed to increase lipid content in animal diets (S9) decreased CC and CC/LULUC per ha but increased CED, EP and AC by 4-10%. The combination of S1, S2, S5, S6 and S8 increased impacts per ha by 2% for CED and 8% for CC, CC/LULUC and AC, but decreased EP by 11%.

3.2. Effects of changes in farming practices on LO and alternative land use on CC/LULUC

Farming practices such as decreasing grass loss on pasture (S2), fattening female calves instead of rearing them as replacement heifers (S4), decreasing calving age (S6) and combination of S1, S2, S5, S6 and S8 (S10) decreased the use of permanent grassland and total land occupation per kg of carcass mass by 12-23% and 9-19%, respectively (Table 4). If Corsican pine were planted on the released permanent grassland, CC/LULUC both per kg of carcass mass and per ha occupied on- and off-farm decreased by 20-27% for S2, S4, S6 and 46-48% for S10 (Table 5). Corsican pine planted on r eleased permanent grassland did not affect CED per either FU.

Table 4: Land occupation (m^2*year/kg carcass mass) of baseline for standard beef-cattle production and farming-practice scenarios*

Land-use type	Baseline**	S1	S2	S3	S4	S5	S6	S7	S8	S9	S10
Permanent pasture	34.0	34.0	29.8	35.7	29.3	33.8	29.8	34.4	34.0	33.9	26.3
Temporary pasture	4.7	4.7	4.1	4.9	4.1	4.7	4.1	3.7	4.6	4.6	3.6
Arable land on-farm	8.3	8.3	8.3	7.3	8.9	8.2	8.2	9.3	8.3	6.4	8.0
Arable land off-farm	1.0	1.0	1.0	1.0	1.2	1.0	1.1	0.7	1.0	2.3	1.0
Other land off-farm	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Total	48.2	48.2	43.4	49.0	43.7	47.7	43.4	48.3	48.1	47.4	38.9

*Scenarios are defined in Material and Methods
**Nguyen et al. (2012a)

Table 5: Area (m^2*yr) of "excess" permanent grassland converted to forest and climate change impact of beef-cattle production including LULUC* (kg CO_2 eq.) as a function of functional unit and farming-practice scenario**

	Baseline***	**S2**	**S4**	**S6**	**S10**
Per kg of carcass mass					
Forest (m^2*yr)	0	4.2	4.7	4.2	7.8
Climate change with LULUC	25.5	20.5	19.2	18.7	13.4
Per ha of land occupied					
Forest (m^2*yr)	0	883	974	885	1666
Climate change with LULUC	5293	4306	3975	3933	2862

* Land use and land-use change
** Scenarios are defined in Material and Methods
***Nguyen et al. (2012a)

4. Discussion

4.1. Differences according to impact category and functional unit

At the scale of the entire beef production system, farming practices for mitigating GHG emissions showed compensation among GHG emissions compared to the baseline scenario. Environmental impacts per kg carcass mass varied according to farming practice, from no effect (S7) to slight decreases (S1, S2 except for a high decrease in EP, S5) or large decreases (S6, S10) of all impacts. In contrast, for S9, CED and EP increased by 7-8%, while other impacts decreased, due to an increase in energy requirements for linseed production, the extrusion process and concentrate production for finishing and cow-calf diets. For S3, LO increased due to an increase in hay for winter feeding, even though cereal use decreased. For S8, CED decreased more than other impacts because rapeseed meal required less energy than soybean meal, mainly due to transport within Brazil and from Brazil to France (Nguyen et al., 2012b).

Most studies of GHG mitigation strategies of ruminant-production systems expressed impacts per kg of animal product because GHG emissions are considered global impacts and their driver is global demand for these products. Environmental impacts can be expressed per ha of land (on-farm and off-farm) occupied, however, if the driver is reducing pollution in a given area (Nguyen et al., 2012b). The relative impacts of some farming practices (S2, S3, S4, S6 and S10) differed according to the FU (kg of carcass mass or ha of land occupied), because these practices had a larger effect on LO than on other impacts. Regarding local impacts, eutrophication and acidification per ha did not increase more than 8-9% among scenarios. However, it is unlikely that potential impacts would reach levels that result in actual eutrophication and acidification damage, especially since the cow-calf herd is based principally on grassland with a moderate stocking rate and a low mineral-fertiliser application rate (Nguyen et al., 2012b).

4.2. Effect of farming practices on impacts

Suckler beef production in France is specialised in suckler cow-calf herds with finishing heifers and bull-fattening herds. Environmental impacts of beef production mainly originate from the cow-calf herd (Pelletier et al., 2010; Beauchemin et al., 2011). Nguyen et al. (2012a) showed that the suckler cow-calf herd with finishing heifers contributed 83-95% of impacts of the whole system. As an example, replacing soybean meal with rapeseed meal in bull diets (S8) had modest effects on the impacts of the whole system, even though it decreased the CC/LULUC and CED impacts of the bull-fattening herd by 9 and 22%, respectively (results not shown). In France and more generally in Europe, suckler cow-calf herds are produced principally on grassland area which is used for grazing in summer and production of conserved forages used in winter. This characteristic limits the ability to modify their diets (Foley et al., 2011; Nguyen et al., 2012a), for example with ingredients (e.g., additives, lipids) that decrease enteric CH_4 production.

Decreasing mineral N fertiliser application to permanent grassland (S1) slightly decreased impacts of the whole system because its use was already low in the baseline. It can, however, reduce production costs. The main advantage of decreasing grass losses on grazing (S2) is a reduction in grassland occupied per kg of beef produced. However, it requires more work to farmers for grassland management, in particular adapting grazing to grass growth by the systematic use of rotational grazing.

Underfeeding heifers in winter (S3) decreased impacts of the whole system little because the reduction in impacts of cereal ingredients was compensated by the increase in those of forages. Although heifers required less net energy for growth, total DM intake increased because digestibility of hay and fresh grass is lower than that of cereals. However, this scenario promotes the use of grassland for ruminant production, which increases C sequestration in soils and avoids using crops that could be fed to humans.

Fattening female calves instead of rearing them as replacement heifers (S4) reduced impacts because they grew faster, resulting in less rearing time before slaughter. Also, their enteric CH_4 emissions were lower as they were fed with maize silage and concentrate instead of mainly forage. As maize silage has a higher yield per ha than grass, the area of grassland used for the herd decreased. Even though this practice increased the use of feed-crops, it can be considered as a potential climate change mitigation practice.

Among farming practices evaluated, decreasing calving age (S6) seems one of the most effective impact-mitigation strategies, as impacts decreased by 8-10% due to two effects. First, all heifers were reared at higher growth rates to reach minimum body condition for first breeding at 15 months and first calving at 24 months instead of 27 and 36 months, respectively. In this way, one year of cow rearing (6 instead of 7 years) was saved without reducing reproductive yield per lifetime. Second, heifers not used for replacement also grew faster, thus finishing sooner (23 instead of 33 months), reducing impacts of the whole production system (as explained for S4). First calving at 2 years is the current practice in western Canada (Beauchemin et al., 2010). In France, first calving at 2 years with the Charolais breed was begun in experimental farms and later implemented by some innovative farmers (Farrié et al., 2008). Changing first calving from 3 to 2 years for half of a Charolais herd improved profit when the number of calvings per cow was increased by 5-10% (Farrié et al., 2008).

Increasing cow longevity (S5) decreased impacts of the whole system little, as the annual number of cull-cows decreased but that of finished heifers increased. Using different allocation methods, Nguyen et al. (2012a) showed that impacts per kg carcass mass of finished heifers slaughtered at 33 months were relatively higher than those of 7-year-old cull cows (except for mass allocation). In S5, impact reductions obtained by extension of cow lifetime were compensated by high impacts of finished heifers. Beauchemin et al. (2011) observed a similar result for GHG emissions and argued that the additional beef produced had higher per-kg GHG emissions. This practice will mitigate impacts more if impacts of finished heifers could be reduced. It is possible that combining this practice with fattening female calves instead of rearing them as replacement heifers (S4) could reduce impacts of the entire system.

Concerning feeding practices, the partial replacement of protein supplement by lucerne hay during the winter (S7) did not affect impacts, as the percentage of protein supplement replaced was small (0.8% of total DM intake of the cow-calf herd) and only 30% of it was soybean meal. Adding lipids to finishing diets to reduce enteric CH_4 production slightly decreases total GHG emissions of beef-production systems (Stewart et al., 2009; Beauchemin et al., 2011; Nguyen et al., 2012a). In this study, adding lipids both to finishing and cow-calf diets (S9) decreased GHG emissions per kg carcass mass by no more than 3%, which was lower than the 11% decrease obtained by Beauchemin et al. (2011). This difference is due to including a lower percentage of lipids in the winter cow-calf diet in this study than in that of Beauchemin et al. (2011) (1.2 *vs*. 4%, respectively).

A combination of several compatible scenarios (S10) appeared the most promising impact-mitigation strategy. Overall, the effects of each farming practice on impacts were limited because each affected only one element of the whole system. In our study, combining several farming practices, even when taking into account known interactions, approximately equalled the sum of the effects of each individual practice. With additional research on system experiments, currently unknown interactions between these practices might be identified that could modify our predictions. Del Prado et al. (2010), comparing a variety of GHG-mitigation options using either a simulation model or an aggregation of single-effect options, found that the aggregation of single-effect options tended to overestimate overall GHG mitigation. We cannot exclude that an overestimation of this type occurred in this study.

The cost of implementing practices was not evaluated. It is obvious that financial costs will influence the implementation of these practices on farms (Beauchemin et al., 2011). Vellinga et al. (2011) observed that farmers tend to choose mitigation options that are relatively simple and either cost-effective or inexpensive. However, to reach a significant effect, the combination of several practices is necessary; this is more challenging for farmers and raises the problem of the farmer acceptability of these practices. One way of making mitigation practices more acceptable to farmers may lie in the attribution of subsidies to offset the cost of these practices.

A "cradle-to-farm gate" LCA study of beef production system requires numerous production parameters, emission factors, empirical equations and modelling assumptions, which can substantially affect the uncertainty of results (Crosson et al., 2011). The uncertainty in estimates of GHG emissions for milk production results mainly from uncertainty in emission factors used to estimate N_2O and enteric CH_4 emissions, DM intake, and milk yield (Basset-Mens et al., 2009; Flysjö et al., 2011; Henriksson et al., 2011). We can assume this holds true for our study, if one substitutes "milk yield" with "animal growth. In our study, enteric CH_4 was estimated with the Tier-3 method used in the French Inventory of Greenhouse Gases, as recommended by IPCC (IPCC, 2006) to improve the accuracy of emission estimates. We consider this estimate to have low uncertainty. Our estimates of N_2O emissions are based on IPCC Tier 2 (IPCC, 2006) emission factors, which have a high uncertainty. Our data on DM intake and animal growth have low uncertainty, as they are based on French feeding system table (INRA, 2007). The scenarios we compared are assessed in the same local conditions as the baseline; thus, variability due to weather and soil conditions is excluded. On the whole, therefore, our scenarios have low uncertainty for the main factors determining GHG emissions of beef production, except for N_2O emissions.

4.3. Alternative land use

This paper explores the potential of even-aged forest as an alternative land use for permanent grassland released due to more efficient farming practices, illustrating its potential for reducing the CC/LULUC impact of the entire farm system when comparing farming practices. Apart from forest, there is no alternative land use for permanent grassland that can increase C sequestration in both soil and biomass. This option appeared the most promising GHG mitigation strategy for the beef production system without altering farm productivity. However, this is a short- and mid-term GHG mitigation strategy, as C sinks resulting from sequestration activities in soil or biomass are not permanent (e.g. Smith, 2005). In our study, the forest is harvested 64 years after planting, which implies a partial return of the C stock in its biomass to the atmosphere. The dynamics of this return of C to the atmosphere will depend on how the biomass will be processed and into what products. A considerable fraction of the harvested biomass may be used for energy production, which may result in a rapid return of C to the atmosphere, unless carbon capture and storage technologies are implemented. In either case, this biomass can replace fossil energy, thereby mitigating GHG emissions and non-renewable energy use. A part of the wood may be used as construction material, resulting in C storage over a longer time period (e.g., 40 years) (Vallet et al., 2009). Furthermore, after harvesting, a new forest can be planted and resume C sequestration. Finally, it is expected that in the long-term, effective technologies and solutions will be achieved for global GHG mitigation; it is therefore crucial to identify effective practices for GHG mitigation in the short- and mid-term.

In practice, planting even-aged forests is both labour-intensive and regulated at regional levels. Although the introduction of even-aged forest in regions dominated by grassland-based bovine production may not be welcomed by all stakeholders concerned, it certainly has a major potential to contribute to short- and mid-term GHG mitigation. In addition, the edge effect among forest, pastures, and cropland may increase biodiversity of the production system (Benton et al., 2003). In crop-farm systems, identifying and simulating alternative land-uses strongly affected their environmental impacts (Tuomisto et al., 2012). Furthermore, comparing farming practices with identical farm area (i.e. considering alternative land-uses on farms) avoids relative changes in impacts according to functional unit (per unit mass or area).

5. Conclusion

It is difficult to greatly reduce the environmental impacts, and in particular the GHG emissions, of a beef-cattle production system, as its impacts result to a large extent from the suckler cow-calf herd; this offers few options to modify herd management and feeding strategies. Modification of individual farming practices moderately affected impacts of the whole beef system; the most promising practice is a decrease in calving age from 3 to 2 years. Our results suggest that simultaneous application of several compatible farming practices can reduce impacts of beef-cattle production significantly. However, our scenario did not consider possible interactions between practices. This point should be further explored, and an approach combining system experiments and simulation modeling seems appropriate. The introduction of even-aged forest as an alternative land-use in beef-cattle farms seems promising and merits further exploration. It illustrates that when comparing farming practices, alternative land-use may strongly affect the climate-change impact of the entire production system.

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Supplementary materials

Table S1: Main inputs used, dry matter yield and nitrate-N emitted (kg/ha, except for Irrigation water in m^3/ha) for pastures and the major feed crops[1]

Pasture or crop type	N mineral	N manure	P_2O_5 (triple superphosphate)	K_2O (potassium chloride)	CaO	Seed	Pesticide (active ingredient)	Diesel	Agricultural machinery	Irrigation water	Plastic for silage	Yield (dry matter)[2]	Nitrate-N emitted
Permanent grassland	28	27	19	39	167	0	0	30	5	0	0	5640	20
Temporary grassland	33	0	28	58	167	6	0	51	13	0	0	8280	20
Lucerne	0	0	119	256	167	8	0.9	128	27	0	0	9000	15
Silage maize	57	138	31	29	167	20	1.0	91	22	354	16	11000	40
Wheat	171	7	37	24	167	140	2.6	99	23	0	0	5650	40
Barley	129	6	37	26	167	125	2.6	100	24	0	0	5550	40
Soybean from Brazil	6	1	80	80	518	53	1.7	76	18	0	0	2708	18
Rapeseed	165	16	50	50	167	3	1.1	92	20	0	0	3040	40
Sunflower	39	27	34	40	167	5	2.0	66	17	0	0	2100	40
Linseed	70	0	45	25	0	46	0.8	90	22	0	0	1800	40

[1]Data for grassland and all crops concern a one-year period, except for soybean, where data are for a six-month period

[2]Yield of grassland corresponds to the yield obtained when all grass is machine harvested. 23% and 75% of the yield of permanent and temporary grassland, respectively, was machine harvested as conserved forage (hay and/or wrapped grass silage). Losses during conservation for both hay and wrapped grass silage were assumed to be 6% of the initial DM. Grass not harvested as conserved forage was available for intake by animals during grazing. For several reasons (selective grazing, trampling of grass, unfavourable weather conditions) a part of the grass grown is not ingested, this “loss” corresponded to 31.5% of grass dry matter available for grazing.

Table S2: Emissions sources, equation or emission factor used and reference

Pollutant/source		Equation/emission factor	Reference
Manure management			
Direct N_2O	= N excreted (kg) x EF[1] x 44/28		IPCC 2006 Tier 2
	deep bedding manure	EF = 0.07 kg N_2O-N/kg N	
	slurry without natural crust cover	EF = 0 kg N_2O-N/kg N	
Indirect N_2O	= N excreted (kg) x $Frac_{Gas}$[2] (%) x 0.01 x 44/28		IPCC 2006 Tier 2
	deep bedding manure	$Frac_{Gas}$ = 30%	
	slurry without natural crust cover	$Frac_{Gas}$ = 40%	
CH_4	= [[GEI[3] x (1-DE[4]%)/100 + UE*GEI[5]] x 0.92)/18.45] x 0.17 x 0.67 x MCF[6] (%)/100		IPCC 2006 Tier 2
	deep bedding manure	UE = 0.04; MCF = 4	
	slurry without natural crust cover	UE = 0.04 for SM and SML and 0.02 for SCL and FC; MCF = 27%	
NH_3	in housing	=0.12 x N excreted (kg) x 17/14	Payraudeau *et al.,* 2007
	in storage	=0.06 x N remaining (kg) x 17/14	
Cropping and grassland production			
Direct N_2O	=[[(mineral N (kg) + liquid N (kg) + cattle manure N (kg) + residue N (kg)] x 0.01 + N deposited by grazing x 0.02)] x 44/28		IPCC 2006 Tier 2
Indirect N_2O	=[[[(mineral N (kg) + liquid N (kg)] x 0.1 + cattle manure N (kg) x 0.2] x 0.01 +N-NO_3 (kg) x 0.0075] x 44/28		IPCC 2006 Tier 2
NO_x	= 0.21 x N_2O (kg)		Nemecek and Kägi, 2007
NH_3	= (0.02 x mineral N (kg) + 0.08 x liquid N (kg) + 0.076 x cattle manure N (kg) + 0.08 x N deposited by grazing) x 17/14		Nemecek and Kägi, 2007; Payraudeau *et al.,* 2007; CORPEN 2006
NO_3	Cropping		Basset-Mens *et al.*, 2007
	Grassland	$= 8.77\ e^{0.003 \text{ x grazing days/ha/LU7}}$ x 62/14	Vertès *et al.,* 1997
P leaching	Cropping	=0.07 kg P/ (ha x yr)	Nemecek and Kägi, 2007
	Grassland	=0.06 kg P/ (ha x yr)	
P run-off	=P run-off lost x [1 + 0.2/80 x mineral P_2O_5 (kg) + 0.4/80 x manure P_2O_5 (kg) + 0.7/80 x P_2O_5 deposited by grazing (kg)]		Nemecek and Kägi, 2007
	Cropping	P run-off lost = 0.175 kg P/ (ha x yr)	
	Grassland	P run-off lost = 0.15 kg P/ (ha x yr)	
P erosion	=10000 x (80 x 0.033 x 0.38 x 0.65 x effect of the vegetation cover factor) x 0.00095 x 1.86 x 0.2 kg P/ (ha x yr)		Nemecek and Kägi, 2007; Nemecek *et al.*, 2003

[1]EF: emission factor for direct N_2O emissions from manure management

[2]$Frac_{Gac}$: % of managed manure nitrogen for production system that volatilises as NH_3 and NO_x

[3]GEI: gross energy intake

[4]DE: digestibility of the feed

[5]UE x GEI: urinary energy expressed as fraction of GEI

[6]MCF: methane conversion factor from each manure-management system (in %)

[7]LU: livestock unit

Chapter 4

Effect of dairy production system, breed and co-product handling methods on environmental impacts at farm level

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Effect of dairy production system, breed and co-product handling methods on environmental impacts at farm level

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Abstract

Six dairy farms with the same on-farm usable agricultural area (UAA) and milk production were compared. One farm (G-No) used grass as the sole forage for a herd of Normande cows, a dual-purpose breed. Three farms, with Holstein cows, varied forage for the herd from grass only (G-Ho) to low (G/LM-Ho) or high (G/HM-Ho) proportion of maize silage in the total forage area. Finally, two farms based on G/LM-Ho and G/HM-Ho systems aimed to increase omega-3 fatty acids in the winter diets of cows (G/LM/O3-Ho, G/HM/O3-Ho). Allocation methods (biophysical, protein, economic allocation) and system expansion applied for co-product (milk and meat) handling were examined. The impact categories considered were climate change, climate change including the effects of land use and land use change (CC/LULUC), cumulative energy demand, eutrophication, acidification and land occupation. The impacts per kg of fat-and-protein-corrected milk (FPCM) or per kg of liveweight (LW) of animals (surplus calves and finished cull cows) of G-No were highest, followed by those of G-Ho, G/LM-Ho and G/HM-Ho, regardless co-product handling methods (except for LW of animals with system expansion) and impact categories (except for eutrophication). CC/LULUC per kg FPCM and per kg LW of G/LM/O3-Ho and G/HM/O3-Ho were both 1% and 3% lower than those of G/LM-Ho and G/HM-Ho, respectively, but other impacts were higher. With system expansion, impacts per kg FPCM were lower than those of allocation methods, but impacts per kg LW of each type of animal were identical among G-No, G-Ho, G/LM-Ho and G/HM-Ho and among G/LM/O3-Ho and G/HM/O3-Ho. Enteric fermentation was the greatest contributor (45-50%) to CC/LULUC, while grass production was the most important contributor to other impacts. The highest CC/LULUC (for G-No) can be explained by (1) G-No having the lowest milk yield/cow (though it produced the most meat) and (2) the fact that grass required more N fertiliser, but had lower yields than silage maize, even though grassland sequestered C. Among Holstein systems, increasing cow productivity by increasing feed intake (including maize silage and supplementing with concentrate) decreased impacts of milk and meat. Reducing replacement rate and age of first calving also decreased impacts of milk and meat. Increasing cow productivity reduced the amount of on-farm UAA required to produce a given amount of milk. Thus, the “liberated” on-farm UAA of Holstein systems was used to produce cash crops, and total impacts of these systems were lower than those of G-No (except for eutrophication and total on-farm and off-farm UAA).

Keywords: dairy farm, grass-based, maize silage, cow breed, co-product handling method, life cycle assessment

1. Introduction

Worldwide, the dairy sector is estimated to contribute 4% (±26%) to total global anthropogenic greenhouse gas (GHG) emissions (FAO, 2010). Methane (CH_4) emissions are the most important contributor (52%) to the climate change impact of milk production, followed by nitrous oxide (N_2O) and carbon dioxide (CO_2) emissions (FAO, 2010). Other environmental impacts related to the dairy sector, such as eutrophication (mainly from nitrate leaching), acidification (from ammonia emissions), land use and energy use depend on farming system (de Vries and de Boer, 2010). Supplementation of cattle diets with lipids rich in omega-3 fatty acids (FAs) from linseed significantly decreased enteric CH_4 emissions from dairy cows (Martin et al., 2008). This feeding practice also contributes to a slight improvement of milk nutritional quality (Chilliard et al., 2009). However, the effectiveness of CH_4 mitigation strategies must be evaluated for entire production systems, and not only for total GHG emissions but also for other environmental impacts (Nguyen et al., 2012) (chapter 2).

Management practices (e.g. breed, feeding strategy, herd management, etc.) of dairy systems in France are highly diverse and depend on farmer goals. We examined dairy farms in a French dairy region (Normandy) using grass as the sole forage for the herd based on Normande cows, a dual-purpose breed, compared with farms based on Holstein cows, a specialised milk-producing breed. In the latter case, forages used for the herd were varied by including different quantities of maize silage and supplemented with concentrate feed to increase milk productivity. We modelled cow diet supplementation with lipids rich in omega-3 FAs to decrease enteric CH_4 produced by cows. The purpose of this study was to evaluate effects of breed, feeding strategies and production systems on the environmental impacts of dairy farms. An additional objective was to evaluate the effects of co-product handling methods on the estimated impacts of milk and its co-products (surplus calves and cull cows).

2. Materials and methods

2.1. Description of dairy farms

Six dairy farms (i.e. a dairy sub-system plus an optional cash-crop system) with the same on-farm usable agricultural area (UAA; 55 ha) and milk production (a quota of 250 000 l milk with 4% fat content) were modelled and compared. The UAA was defined according to the dairy sub-system requiring the largest forage area for its herd (Delaby and Pavie, 2008). For the other farms, the UAA not used for forage production was available for cash crops (i.e. wheat and rapeseed). Wheat was introduced to supply straw for bedding, and rapeseed was used to complete the rotation with grassland. These dairy sub-systems were characterised by the proportion of maize silage in the total forage area (three levels: “grass only”, “low” or “high”), cow breed (Holstein or Normande) and whether or not cow rations were supplemented with omega-3 FAs during the non-grazing period. For the 3 farms with low maize-silage proportion and Holstein (G/LM-Ho), grass only and Holstein (G-Ho) and grass only and Normande (G-No), data for these systems obtained in an experimental farm provided data for cow diets and cow productivity (Delaby et al., 2009), herd characteristics and total UAA (Delaby and Pavie, 2008). For the farm with high maize-silage proportion and Holstein (G/HM-Ho), a dairy case-study in Normandy representing a typical high maize-silage dairy farm provided data for cow diets, cow productivity and herd characteristics (Pavie et al., 2010), which were adjusted to conform to the milk production, UAA and grassland productivity of the other three farms. Substitution of a portion of concentrates with extruded linseed (a source of omega-3 FAs) was developed as alternate diets for the two farms with maize silage and Holstein (G/LM/O3-Ho and G/HM/O3-Ho). Because these dairy farms were designed to represent real farms, herd size, herd management and on-farm cropping pattern differed from one system to another.

Animal production

The G/HM-Ho dairy sub-system had 33% of silage maize in the forage area with highly productive Holstein (8.66 t Fat and Protein Corrected Milk (FPCM)/cow/yr). The herd consisted of 32 cows (11 primiparous) that annually provided 29 calves, 13 of which were kept to be raised as heifers to replace cull cows. Cows were housed from September to mid-March and mainly fed maize silage (16.0 kg dry matter intake (DMI)/d) and concentrate (5.0 kg DMI/d). During the grazing (including transition) period (120 days), cows mainly grazed and were supplemented with maize silage (4.5 kg DMI/d) and concentrate (1.1 kg DMI/d).

The G/LM-Ho dairy sub-system had 11% silage maize in the forage area with highly productive Holstein (8.17 t FPCM/cow/yr). The herd consisted of 32 cows (14 primiparous) that annually provided 32 calves, 16 of which were raised as heifers to replace cull cows. Cows were housed from mid-December to the end of March and fed maize silage (13.4 kg

DMI/d) and concentrate (6.3 kg DMI/d). During the grazing (including transition) period (205 days), cows mainly grazed and were supplemented with concentrate (4.0 kg DMI/d) and maize silage (1.6 kg DMI/d).

Two G-Ho and G-No dairy sub-systems were based on grassland only (i.e. no silage maize in the forage area) and used Holstein (6.74 t FPCM/cow/yr) or Normande (5.72 t FPCM/cow/yr), respectively. Respectively, the herds consisted of 41 and 49 cows (17 and 15 primiparous) that annually provided 38 and 46 calves, 19 and 23 of which were raised as heifers to replace cull cows. In these systems, cows were housed from mid-December to the end of March and mainly fed conserved grass (i.e. silage, big bale silage and hay) and concentrate (2.5 kg DMI/d). During the grazing (including transition) period (205 days), cows mainly grazed and were supplemented with silage (1.6 kg DMI/d) and minerals (0.4 kg DMI/d).

Herd characteristics and management of the two dairy sub-systems with diets enriched in omega-3 FAs (G/HM/O3-Ho and G/LM/O3-Ho) were the same as those of G/HM-Ho and G/LM-Ho, respectively. Only the concentrate fed to lactating cows during the winter and the transition period was modified by including extruded linseed to reach about 2% FAs added per kg DMI, but total energy and protein in diets were the same as those of G/HM-Ho and G/LM-Ho systems. In the absence of published results documenting consistent changes in milk yield or fertility when omega-3 FAs were given (Petit, 2010), we assumed that milk yield per cow did not change.

Cows in the G/HM-Ho and G/HM/O3-Ho systems had their first calving at 25 months, all calving was grouped at the end of summer and drying off took place on pasture supplemented with maize silage and concentrate. Each heifer from 0-25 months consumed (in DMI) about 3.4 t grazed grass, 1.6 t grass silage and hay, 0.5 t maize silage and 0.4 t concentrate. For the other systems, cows had their first calving at 36 months, all calving was grouped at the end of autumn and drying off took place during the indoor period with grass silage. Each heifer from 0-36 months consumed (in DMI) about 4.3 t grazed grass, 3.4 t DMI grass silage and hay and 0.5 t concentrate.

For each dairy sub-system, it was assumed that 8% of the milk produced was not commercialised because it was fed to calves or lost to diseases such as mastitis. Calves not kept for replacement were sold after 2 weeks (55 kg live weight, LW). We assumed that one heifer died at the end of the first year. Heifers were raised up to their first calving, but those which were not incorporated into the cow herd were sold as pregnant heifers. It was assumed that one cull-cow died; the rest were fattened with grass silage and concentrate (2 kg DMI/d) for two months before slaughter. Extruded linseed was included in concentrate for cull cows of G/HM/O3-Ho and G/LM/O3-Ho, but total energy and protein in diets were the same as for those of G/HM-Ho and G/LM-Ho systems. We assumed that losses during grazing or conservation and feeding of grass were about 15%, and losses during conservation and feeding of maize silage were about 10%. Table 1 describes the six dairy production systems.

Cows were raised indoors in a loose housing system, and slurry from the feeding area was evacuated and stored outside the animal housing. The bedding area for cows was covered with straw, and solid manure was collected indoors and removed every 2 months. Other types of animals were housed in deep bedding, and manure was collected indoors and removed once a year. Impacts associated with the construction and maintenance of cattle housing, forage and manure storage were included, based on N guyen et al. (2012). The use of machines and energy for milking, housing illumination, feeding, mulching, carrying manure out of housing and cleaning were included, based on data from Dollé et al. (2009).

Table 1. Technical description of the G-No, G-Ho, G/LM-Ho, G/HM-Ho, G/LM/O3-Ho and G/HM/O3-Ho dairy farms[1] analysed

		G-No	G-Ho	G/LM-Ho	G/HM-Ho	G/LM/O3-Ho	G/HM/O3-Ho
On-farm area	ha	55.0	55.0	55.0	55.0	55.0	55.0
One-year leys	ha	4.0	3.5	2.1	0	2.1	0
Temporary grassland	ha	22.8	19.6	12.2	9.3	12.	9.3
Permanent grassland	ha	28.2	24.6	17.5	10.0	17.5	10.0
Maize silage	ha	0	0	4.1	9.4	4.1	9.4
Wheat	ha	0	7.4	15.1	17.8	15.1	17.8
Rapeseed	ha	0	0	4	8.5	4	8.5
Herd characteristics							
Cows	number	49	41	34	32	34	32
Heifers (0-1 yr)	number	23	19	16	13	16	13
Heifers (1-2 yr)	number	22	18	15	12	15	12
Heifers (2-3 yr)	number	22	18	15	0	15	0
Replacement rate	%	31	41	41	34	41	34
Cull rate	%	29	39	38	31	38	31
FPCM[2]	t/cow/yr	5.72	6.74	8.17	8.66	8.17	8.66
Total herd requirements							
Grass	t DM[3]	266.3	224.1	186.1	129.2	186.1	129.2
Hay	t DM	36.9	35.2	33.3	12.4	33.3	12.4
Grass silage	t DM	150.3	131.0	64.2	28.7	64.2	28.7
Bale silage	t DM	38.0	34.0	0	0	0	0
Maize silage	t DM	0	0	61.3	140.7	61.3	140.7
Wheat	t DM	15.6	13.2	39.1	11.4	32.4	2.3
Soybean meal	t DM	10.4	8.8	19.8	29.1	18.0	25.6
Extruded linseed	t DM	0	0	0	0	6.4	11.2
Minerals	t DM	5.2	4.4	3.0	3.9	3.2	4.1
Straw for bedding	t DM	76.3	63.3	52.6	37.9	52.6	37.9
Dairy farm outputs (sold products)							
FPCM[2]	t	258.0	254.2	255.6	254.9	255.6	254.9
Finished cull cows	t LW[4]	10.8	11.8	9.6	7.4	9.6	7.4
Weaned calves	t LW	1.3	1.1	0.9	0.9	0.9	0.9
Pregnant heifers	number	7	1	1	1	1	1
Wheat	t DM	0	47.2	96.3	113.5	96.3	113.5
Wheat straw	t DM	0	0	5.1	30.2	5.1	30.2
Rapeseed	t DM	0	0	12.9	27.4	12.9	27.4

[1]Dairy farms are defined in Materials and Methods

[2]Fat and protein corrected milk

[3]Dry matter

[4]Live weight

Grassland and crop production

Grassland area consisted of permanent grassland or temporary or one-year leys sown with perennial ryegrass (PRG) or PRG with white clover. Permanent grassland was used either for grazing only (by cows or heifers) or for conserved forages (i.e. silage, baled silage or hay) only. Permanent grassland had no tilling or sowing operations. Temporary grassland was renewed every five years in average. Temporary grassland was grazed,and surplus grass in spring and summer was harvested for conserved forages. Leys were renewed annually and only harvested as conserved forage. For G/HM-Ho and G/HM/O3-Ho, total grass area for grazing was 32 ares/cow/yr and 58 ares/heifer from 0-2 years. For the other dairy sub-systems, these total grass area was 44 ares/cow/yr and 70 ares/heifer from 0-3 years. Grassland received mineral nitrogen (N) fertiliser and manure (stored in winter or deposited from grazing). Annual grassland practices, inputs and yields are summarised in Table S1.

Silage maize yielded 15 t dry matter (DM)/ha and received 30 m^3 slurry and 12 t solid manure per ha. In G/LM-Ho and G/LM/O3-Ho, silage maize was always sown after temporary grassland; so, mineral fertiliser requirements were low (18 kg N and 46 kg phosphorus (P) per ha). As in G/HM-Ho and G/HM/O3-Ho, silage maize (sown after temporary grassland or wheat) received on average 65 kg N and 46 kg P/ha/year. Wheat received 120 kg mineral N fertiliser and 8 t solid manure per ha, and it yielded 6.4 t DM of grain and 3.8 t DM straw/ha. Mineral fertilisers supplied rapeseed with 180 kg N, 40 kg P and 60 kg potassium (K)/ha, and it yielded 3.2 t DM/ha. Data for extruded linseed and soybean meal production were based on Nguyen et al. (2012).

Nutritional composition of feed and animal products

Nutritional compositions of feed were estimated based on French data tables (INRA, 2007). Nutritional energy content of milk was estimated based on its fat, protein and lactose contents, for which energy values are 9.1, 5.7 and 3.95 kcal/g, respectively. Nutritional energy content of animals was estimated based on their edible proportions (i.e. meat, 0.43 kg edible product/kg LW, de Vries and de Boer, 2010). It was assumed that nutritional energy content is about 1500 kcal/ kg meat (i.e. edible product). Protein content in animals was estimated based on CORPEN (2001).

2.2. System definition

System boundary and delimitations

This is a cradle-to-farm-gate attributional Life Cycle Assessment study for a one-year period: the studied system includes the production and delivery of inputs used for grassland and crops produced on-farm, feed and bedding produced off-farm, herd management and associated upstream processes, emissions from animals, and manure storage and application to grassland and cropland (Fig. 1). The transport of wheat grain, rapeseed and wheat straw leaving the system and the transport and slaughter of animals leaving the system are not included. Buildings are included, but veterinary medicines are not included due to lack of data.

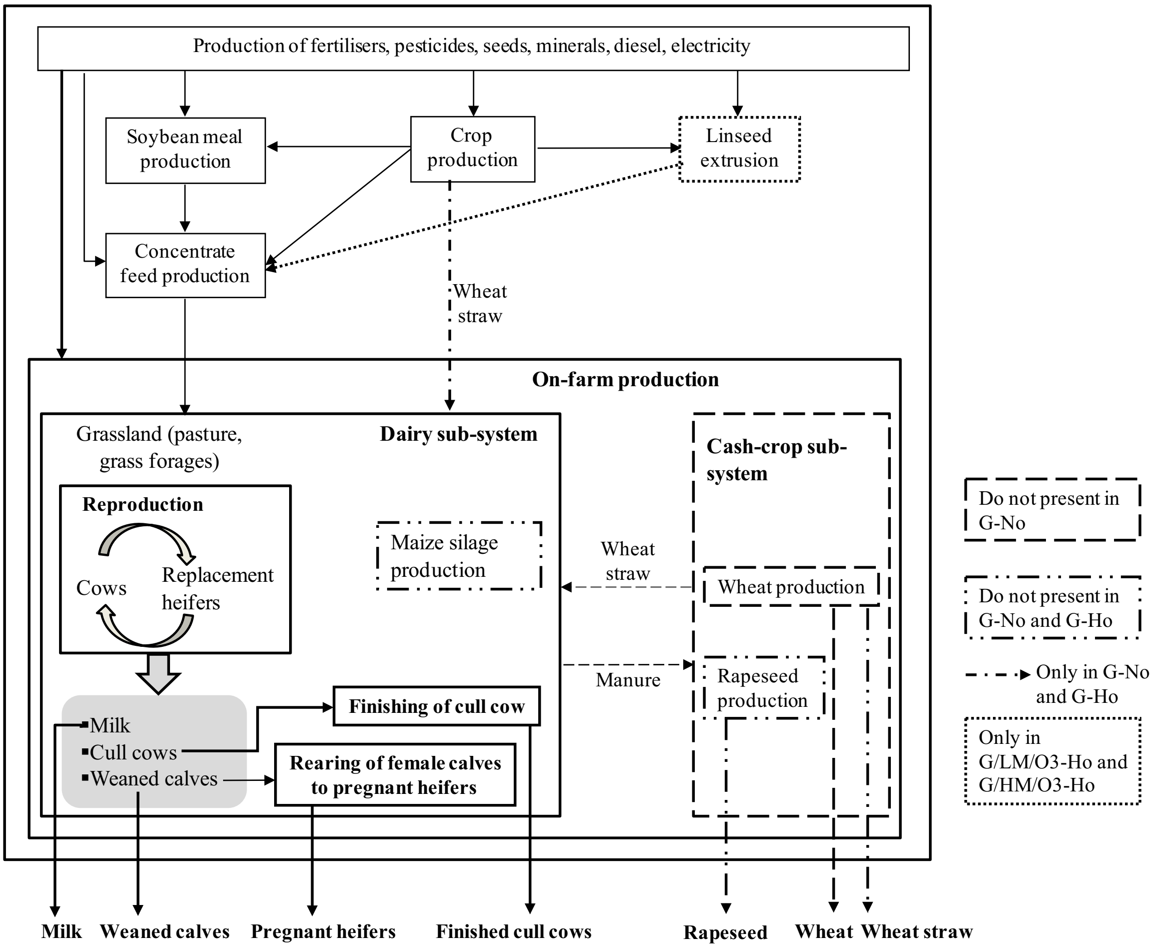

Figure 1. "Cradle to farm-gate" life cycle of G-No, G-Ho, G/LM-Ho, G/HM-Ho, G/LM/O3-Ho and G/HM/O3-Ho dairy farms[1]. On-farm production includes dairy sub-system and cash-crop sub-system (except for G-No). Allocation methods were applied for these products of the common phase (grey shaded rectangle), i.e. the reproduction component, before animals pass to subsequent production stage

[1]Dairy sub-systems are defined in Materials and Methods

Functional unit and co-product handling

The impacts of the dairy sub-system (i.e. whole farm minus cash crops) were attributed to animal products per 1 kg of FPCM and per 1 kg of LW of finished cull cow, weaning calf and pregnant heifer at the farm exit gate. The impacts of cash-crop sub-systems were attributed to crop products (i.e. wheat grain, wheat straw and rapeseed). Economic allocation was used for crop products and feed ingredients resulting from processes yielding several co-products (e.g. wheat grain vs. straw). Methods for co-product handling were applied for animal products (i.e. commercialised milk, cull cows before finishing and weaned calves not used for replacement) produced by the dairy sub-system, which included only dairy cows and replacement heifers. We compared four methods for animal co-product handling (Table 2):

- Biophysical allocation, based on feed-energy requirements needed to produce milk and animals (cull cows and surplus calves; IDF (2010)).
- Allocation according to protein mass, based on the protein content of commercialised milk and the LW of cull cows and surplus calves (CORPEN, 2001).
- Economic allocation, based on the market value of commercialised milk and the LW of cull cows and surplus calves. Prices of these animal products on 2006 w ere calculated from Delaby and Pavie (2008).
- System expansion. The impacts associated with cull cows and surplus calves were subtracted from the total impacts of the dairy herd. Impacts of animals (expressed per kg LW) produced from the dairy herd were assumed to be identical to those of animals delivered from the reproduction component of a beef-cattle production system (i.e. cull cows, surplus calves and breeding bull) (Nguyen et al., 2012).

Table 2. Allocation factors for milk, surplus calves and cull cows according to biological, protein and economic allocations in the G-No, G-Ho, G/LM-Ho, G/HM-Ho, G/LM/O3-Ho and G/HM/O3-Ho dairy sub-systems[1]

	G-No	G-Ho	G/LM-Ho	G/HM-Ho	G/LM/O3-Ho	G/HM/O3-Ho
Biophysical allocation						
% milk	74.0	72.8	77.9	82.5	77.9	82.5
% surplus calves	3.7	2.5	2.1	2.1	2.1	2.1
% cull cows	22.2	24.7	20.0	15.4	20.0	15.4
Protein allocation						
% milk	82.9	81.9	85.1	87.1	85.1	87.1
% surplus calves	2.9	2.0	1.7	1.8	1.7	1.8
% cull cows	14.2	16.1	13.2	11.0	13.2	11.0
Economic allocation						
% milk	80.8	84.9	87.5	89.5	87.5	89.5
% surplus calves	7.8	3.9	3.3	3.4	3.3	3.4
% cull cows	11.4	11.2	9.2	7.1	9.2	7.1

[1]Dairy sub-systems are defined in Materials and Methods

2.3. Emissions estimates

Enteric CH_4 emissions were estimated for each class of cattle according to Vermorel et al. (2008) using animals' net-energy requirements, converted into metabolisable energy intake (MEI) and conversion factors from MEI to CH_4 energy. To represent the effect of diets supplemented with lipids rich in omega-3 FAs, enteric CH_4 production (g CH_4/kg DM intake) was decreased by 4.8% per percentage of added lipids (Martin et al., 2010). Emissions from slurry and solid manure produced by cattle during grazing, in housing and during storage were included as part of the dairy sub-system. Emissions from application of slurry and solid manure on cropland were included as part of the cash-crop production sub-system. Nitrogen excretion was calculated as the difference between an animal's total N intake in feed and the N retained for milk production and growth (meat production). Emissions from slurry and solid manure produced by the herd and from manure application on cropland and grassland were estimated according to IPCC (2006) Tier 2 (for CH_4 and N_2O), CORPEN (2006) and Payraudeau et al. (2007) (for NH_3). Nitrate leaching was estimated based on Vertès et al. (2007) for grassland which was only used for grazing and on Vertès et al. (2012) for other types of grassland and cropland in rotation with temporary grassland. For P excreted on pasture, our estimate was based on CORPEN (1999), taking into account the number of livestock units, the duration of grazing per ha of grassland and milk yield per cow. P emissions (leaching, run-off and erosion) were estimated according to Nemecek and Kägi (2007).

Carbon (C) sequestration for permanent grassland (i.e. older than 30 years) was estimated at 0.7 t CO_2/ha/yr (Arrouays et al., 2002). Except for G-No, temporary grassland was maintained for five years and followed by an annual crop for two years. It was assumed that temporary grassland sequestered about 1.8 t CO_2/ha/yr. When temporary grassland was renewed or replaced by annual crop, C released due to tilling was estimated at 3.7 t CO_2/ha/yr (Arrouays et al., 2002). So, C sequestration for the entire grassland and cropland rotation (5 years of grass, 2 years of annual crops) was estimated at 1.8 t CO_2/ha. For G-No, it was estimated that 50% of temporary grassland was renewed by tilling and 50% by non-tillage techniques (Agreste, 2010); here C sequestration was about 1.5 t CO_2/ha/yr. We assumed that other annual-crop area (including one-year leys) had been converted from permanent grassland more than 20 years ago, and agricultural practices for these crops no longer had an effect on soil C. The proportion of Brazilian soybean crops grown on land converted the previous year from Brazilian rain forest was estimated at 0.7% (Prudêncio da Silva et al., 2010). To conform better to current practice regarding the effect of land-use change on C emissions due to conversion of Brazilian forest to cropland, we adopted a value of 740 t CO_2/ha, as recommended in PAS 2050 (2008), instead of the value of 120 t CO_2/ha used in Prudencio da Silva et al. (2010) and the Ecoinvent database (Jungbluth et al., 2007). The latter estimate corresponds to the estimated of 20% of the above-ground biomass which is burnt, but the remaining 80% is ignored, the reason for this is not specified (Prudêncio da Silva, 2011).

2.4. Life cycle impact assessment

The impact categories considered were climate change (CC), climate change including the effects of land use and land use change (LULUC) (kg CO_2 equivalent (eq.)), cumulative energy demand (CED) (MJ), eutrophication (EP) (g PO_4^{3-} eq.), acidification (AC) (g SO_2 eq.) and land occupation (LO) (m^2*yr). The indicator value for each impact category was determined by multiplying the aggregated resources used and the aggregated emissions of each individual substance with a characterisation factor for each impact category to which it may potentially contribute. The impacts CC, EP, AC and LO were calculated using the CML2 "baseline" and "all categories" 2001 characterisation methods as implemented in the ecoinvent v2.0 d atabase. The CC impact excludes C sequestration in grassland and C emissions due to conversion of Brazilian forest to cropland, whereas the CC/LULUC impact includes them. Total CED was calculated according to version 1.05 of the method, as implemented in the ecoinvent v2.0 database.

3. Results

3.1. Impacts of grassland and other feed ingredient production

Environmental impacts (per ha) for different types of grassland (related to grassland production and forage conservation) varied from 2.9-6.2 t CO_2 eq., 1.7-5.5 t CO_2 eq., 8.0-26.7 GJ, 11.0-33.7 kg PO_4^{3-} eq., 31.6-69.7 kg SO_2 eq. for CC, CC/LULUC, CED, EP and AC, respectively (Table S2). Enteric methane produced by animals during grazing was not accounted for in grassland production, but in animal production. Overall, for all dairy sub-systems, permanent grassland used for cow grazing had highest impacts for CC, CC/LULUC and EP due to the high quantity of N fertiliser used and the long grazing season that increased nitrate leaching. Impacts per ha of grassland only cut for conserved forages were highest for CED due to harvest and forage-conservation operations but lowest for EP due to the lowest nitrate emissions. Temporary PRG with clover had lower CC and CC/LULUC impacts than temporary pure PRG due to less N fertiliser used. Impacts of other feed ingredients expressed per kg DMI are presented in Table S3. Maize silage used in G/LM-Ho and G/LM/O3-Ho had lower impacts than that used in G/HM-Ho and G/HM/O3-Ho, except for EP, because all maize silage was planted after temporary grassland in the former group. Maize silage consequently required less N fertiliser but caused higher nitrate leaching than that in the latter group, where some of it was planted after wheat.

3.2. Impacts of milk and sold animals

When comparing cow breeds, impacts per kg of FPCM of G-Ho were lower than those of G-No by 3-8% (according to the impact category) with biophysical and protein allocations, by 8-26% with system expansion and similar (-2 to +3%) with economic allocation (Table 3). Between maize silage-based and grass-based production systems, impacts per kg FPCM of G-Ho were higher than those of G/LM-Ho by 1-18% and G/HM-Ho by 5-38% with biophysical, protein and economic allocations (except for EP, which was 1-8% lower for G-Ho than for G/LM-Ho and G/HM-Ho). With system expansion, CC/LULUC, EP and LO per kg FPCM of

G-Ho were lower than those of G/LM-Ho by 4-19%; the inverse was the case with CC, CED and AC (1-22% higher). With system expansion, impacts per kg FPCM of G-Ho were higher than those of G/HM-Ho by 1-20%, except for EP (22% lower). With addition of omega-3 FAs in the winter diets, impacts per kg FPCM of G/LM/O3-Ho were higher than those of G/LM-Ho by 0-6%, except for CC and CC/LULUC (1-3% lower) and LO (30% higher, but only with system expansion), and those of G/HM/O3-Ho were higher than those of G/HM-Ho by 1-13%, except for CC and CC/LULUC (1-3% lower) and LO (68% higher, only with system expansion), regardless of co-product handling methods. Per kg FPCM, impacts were highest with protein allocation for G-No and with economic allocation for other systems (with Holstein). Compared to the allocation methods (biophysical, protein and economic), system expansion resulted in the lowest impacts per kg FPCM for each system, but the degree of reduction differed according to impact category and dairy sub-system.

Table 3. Environmental impacts per kg of fat and protein corrected milk from the G-No, G-Ho, G/LM-Ho, G/HM-Ho, G/LM/O3-Ho and G/HM/O3-Ho dairy sub-systems[1] according to four methods for co-product handling

	G-No	G-Ho	G/LM-Ho	G/HM-Ho	G/LM/O3-Ho	G/HM/O3-Ho
Biophysical allocation						
Climate change, kg CO_2 eq.	1.45	1.33	1.29	1.17	1.28	1.16
Climate change/LULUC[2], kg CO_2 eq.	1.34	1.28	1.29	1.22	1.27	1.19
Cumulative energy demand, MJ eq.	4.29	3.97	3.93	3.75	3.98	3.87
Eutrophication, g PO_4^{3-}eq.	4.68	4.37	4.76	4.71	4.92	5.05
Acidification, g SO_2 eq.	12.09	11.26	9.88	9.85	9.93	9.98
Land occupation, m^2*yr	1.46	1.42	1.32	1.10	1.39	1.25
Protein allocation						
Climate change, kg CO_2 eq.	1.62	1.49	1.41	1.24	1.39	1.22
Climate change/LULUC[2], kg CO_2 eq.	1.50	1.44	1.41	1.28	1.39	1.26
Cumulative energy demand, MJ eq.	4.81	4.46	4.29	3.97	4.35	4.09
Eutrophication, g PO_4^{3-}eq.	5.24	4.92	5.20	4.98	5.37	5.34
Acidification, g SO_2 eq.	13.54	12.67	10.79	10.41	10.84	10.54
Land occupation, m^2*yr	1.64	1.59	1.44	1.17	1.52	1.32
Economic allocation						
Climate change, kg CO_2 eq.	1.58	1.55	1.45	1.27	1.43	1.26
Climate change/LULUC[2], kg CO_2 eq.	1.46	1.49	1.44	1.32	1.43	1.30
Cumulative energy demand, MJ eq.	4.69	4.63	4.41	4.08	4.47	4.20
Eutrophication, g PO_4^{3-}eq.	5.10	5.10	5.34	5.11	5.52	5.48
Acidification, g SO_2 eq.	13.19	13.14	11.09	10.69	11.14	10.84
Land occupation, m^2*yr	1.60	1.65	1.48	1.20	1.56	1.35
System expansion						
Climate change, kg CO_2 eq.	1.14	0.97	0.96	0.87	0.94	0.85
Climate change/LULUC[2], kg CO_2 eq.	1.08	1.00	1.03	0.98	1.01	0.95
Cumulative energy demand, MJ eq.	4.01	3.58	3.52	3.35	3.49	3.41
Eutrophication, g PO_4^{3-}eq.	3.37	2.92	3.60	3.72	3.83	4.16
Acidification, g SO_2 eq.	11.48	10.39	8.55	8.67	8.52	8.75
Land occupation, m^2*yr	0.41	0.30	0.36	0.28	0.47	0.47

[1]Dairy sub-systems are defined in Materials and Methods
[2]Climate change including effect of land use and land-use change

The impacts per kg LW of weaned calf and finished cull cow of G-Ho were lower than those of G-No by 1-7% with biophysical and protein allocation (except LO) and by 9-31% with economic allocation (Table 4). With the three allocation methods, the impacts per kg LW of weaned calf and finished cull cow of G/HM-Ho and G/LM-Ho were lower than those of G-Ho by 3-18% and 6-31%, respectively (except for EP). With the three allocation methods, the impacts per kg LW of weaned calf and finished cull cow of G/LM/O3-Ho and G/HM/O3-Ho were higher by 1-7% and 1-13% than those of G/LM-Ho and G/HM-Ho, respectively, except for CC and CC/LULUC (1-2% lower). With system expansion, there were no differences in impacts per kg LW of weaned calf and finished cull cow between G-No, G-Ho, G/LM-Ho and G/HM-Ho, and they were 1-6% lower than those of G/LM/O3-Ho and G/HM/O3-Ho, except for EP and LO. With all methods for co-product handling, there were no differences in impacts per kg LW of pregnant heifer between G-Ho, G/LM-Ho and G/LM/O3-Ho; however, impacts of these treatments were 3-4% higher than those of G-No (except for economic allocation) and 9-28% higher than those of G/HM-Ho and G/HM/O3-Ho (see Table S2). The impacts per kg LW of pregnant heifer were highest (compared to those for weaned calf and cull cow) with biophysical and protein allocation and lowest with system expansion. All impacts per kg LW of finished cull cow were lowest with protein and economic allocation and highest with system expansion. Compared to biophysical allocation, the impacts per kg LW of animals in each system were lower with protein allocation and higher with system expansion.

Table 4. Environmental impacts per kg live weight of weaned calf and finished cull cow from the G-No, G-Ho, G/LM-Ho, G/HM-Ho, G/LM/O3-Ho and G/HM/O3-Ho dairy sub-systems[1] according to four methods for co-product handling

	G-No		G-Ho		G/LM-Ho		G/HM-Ho		G/LM/O3-Ho		G/HM/O3-Ho	
	Weaned calf	Finished cull cow	Weaned calf	Finished cull cow	Weaned calf	Finished cull cow	Weaned calf	Finished cull cow	Weaned calf	Finished cull cow	Weaned calf	Finished cull cow
Biophysical allocation												
Climate change, kg CO_2 eq.	11.3	12.3	10.5	11.7	9.5	10.8	8.2	9.5	9.5	10.7	8.1	9.4
Climate change/LULUC[2], kg CO_2 eq.	10.4	11.5	10.1	11.3	9.5	10.8	8.5	9.8	9.4	10.6	8.4	9.6
Cumulative energy demand, MJ eq.	33.5	36.6	31.5	34.9	29.1	32.7	26.3	30.1	29.5	33.2	27.1	31.0
Eutrophication, g PO_4^{3-}eq.	36.5	37.2	34.7	35.6	35.2	36.1	33.0	34.0	36.4	37.6	35.4	36.6
Acidification, g SO_2 eq.	94.3	97.1	89.3	92.7	73.2	77.8	68.9	73.9	73.5	78.4	69.9	75.0
Land occupation, m^2*yr	11.4	12.4	11.2	12.2	9.8	10.9	7.7	9.0	10.3	11.6	8.7	10.1
Protein allocation												
Climate change, kg CO_2 eq.	8.7	8.6	8.3	8.3	7.6	7.8	7.1	7.4	7.6	7.7	7.0	7.3
Climate change/LULUC, kg CO_2 eq.	8.0	8.1	8.0	8.1	7.6	7.8	7.4	7.6	7.5	7.7	7.2	7.4
Cumulative energy demand, MJ eq.	25.8	25.5	24.9	24.9	23.3	23.7	22.8	23.3	23.6	24.1	23.4	24.0
Eutrophication, g PO_4^{3-}eq.	28.1	25.0	27.4	24.5	28.2	25.1	28.6	25.4	29.1	26.3	30.6	27.4
Acidification, g SO_2 eq.	72.7	65.7	70.6	64.2	58.5	55.1	59.7	56.0	58.8	55.5	60.5	56.8
Land occupation, m^2*yr	8.8	8.6	8.9	8.7	7.8	7.9	6.7	7.0	8.2	8.4	7.6	7.8
Economic allocation												
Climate change, kg CO_2 eq.	23.5	7.3	16.2	6.4	14.9	6.1	12.9	5.5	14.8	6.0	12.7	5.4
Climate change/LULUC, kg CO_2 eq.	21.7	6.9	15.6	6.2	14.9	6.0	13.4	5.6	14.7	5.9	13.1	5.5
Cumulative energy demand, MJ eq.	69.8	21.6	48.5	19.1	45.6	18.3	41.3	17.2	46.2	18.6	42.5	17.7
Eutrophication, g PO_4^{3-}eq.	76.0	20.9	53.5	18.2	55.2	18.6	51.8	17.7	57.0	19.6	55.5	19.2
Acidification, g SO_2 eq.	196.5	54.9	137.8	47.8	114.6	41.6	108.3	39.9	115.2	41.9	109.7	40.5
Land occupation, m^2*yr	23.8	7.2	17.3	6.6	15.3	6.1	12.1	5.2	16.1	6.5	13.7	5.8
System expansion												
Climate change, kg CO_2 eq.	18.2	18.7	18.2	18.7	18.2	18.7	18.2	18.7	18.4	18.8	18.4	18.8
Climate change/LULUC, kg CO_2 eq.	16.1	16.8	16.1	16.8	16.1	16.8	16.1	16.8	16.3	17.0	16.3	17.0
Cumulative energy demand, MJ eq.	39.7	42.3	39.7	42.4	39.7	42.4	39.7	42.4	42.1	44.9	42.1	44.9
Eutrophication, g PO_4^{3-}eq.	65.5	63.9	65.5	63.9	65.5	63.9	65.5	63.9	64.9	63.8	64.9	63.8
Acidification, g SO_2 eq.	107.9	109.7	107.9	109.7	107.9	109.7	107.9	109.7	110.4	112.3	110.4	112.3
Land occupation, m^2*yr	34.9	34.0	34.9	33.9	34.9	33.9	34.9	33.9	34.3	33.6	34.3	33.6

[1]Dairy sub-systems are defined in Materials and Methods
[2]Climate change including effect of land use and land-use change

3.3. Impacts of dairy sub-systems and whole farms

For the dairy sub-system the impacts of G-Ho were 13-17% lower than those of G-No (Table 5). Among Holstein-cow-based dairy sub-systems, impacts were lower by 6-17% for G/LM-Ho and by 6-31% for G/HM-Ho compared to G-Ho, except for EP (only for G/LM-Ho). Impacts of the G/LM/O3-Ho and G/HM/O3-Ho dairy sub-systems were 1-5% and 1-12% higher than those of G/LM-Ho and G/HM/O3-Ho, respectively, except for CC and CC/LULUC (1-2% lower).

For each dairy sub-system, enteric fermentation was the greatest contributor (42-46%) to CC and CC/LULUC, followed by grassland production (17-42%), other feed production (3-14%) and manure management (8-15%) (Fig. 2). Maize silage production contributed 3% to CC and CC/LULUC for G/LM-Ho and G/LM/O3-Ho and 9% for G/HM-Ho and G/HM/O3-Ho. Grassland production contributed 54% to CED for G-No and G-Ho, 35-36% for G/LM-Ho and G/LM/O3-Ho and 19% for G/HM-Ho and G/HM/O3-Ho. Other feed production contributed 13% to CED for G-No and G-Ho, 29-30% for G/LM-Ho and G/LM/O3-Ho and 35-36% for G/HM-Ho and G/HM/O3-Ho. For each dairy sub-system, indoor operations for animals (e.g. milking, feed distribution, manure handling) contributed 28-31% to CED. Grassland production was the largest contributor to EP (47-69%), AC (47-61%) and LO (47-90%) for each dairy sub-systems (except for EP and AC of G/HM-Ho and G/HM/O3-Ho). Manure management contributed 18-22% to EP and 34-43% to AC. Maize silage production for dairy sub-systems of G/LM-Ho, G/LM/O3-Ho, G/HM-Ho and G/HM/O3-Ho contributed 13-30% to EP, 7-18% to AC and 8-26% to LO.

Besides milk and surplus animals, whole farms also produced wheat (except for G-No), wheat straw and rapeseed (except for G-No and G-Ho) (Table 1). Impacts of G-No per whole farm were the highest, except for EP and LO (Table 5). Respectively, the impacts per whole farm of G/LM-Ho and G/HM-Ho were lower than those of G-Ho for CC (2 and 11%), CC/LULUC (5%, only for G/HM-Ho) and AC (12 and 15%) and higher for CED (5 and 4%), EP (20 and 25%) and LO (8 and 4%). The impacts per whole farm of G/LM/O3-Ho and G/HM/O3-Ho were 1-4% and 1-7% higher than those of G/LM-Ho and G/HM-Ho, respectively, except for CC and CC/LULUC (decreased 1%). Cash-crop production contributed to impacts of the whole farm by 3-11% for G-Ho, 8-28% for G/LM-Ho and G/LM/O3-Ho and 12-41% for G/HM-Ho and G/HM/O3-Ho.

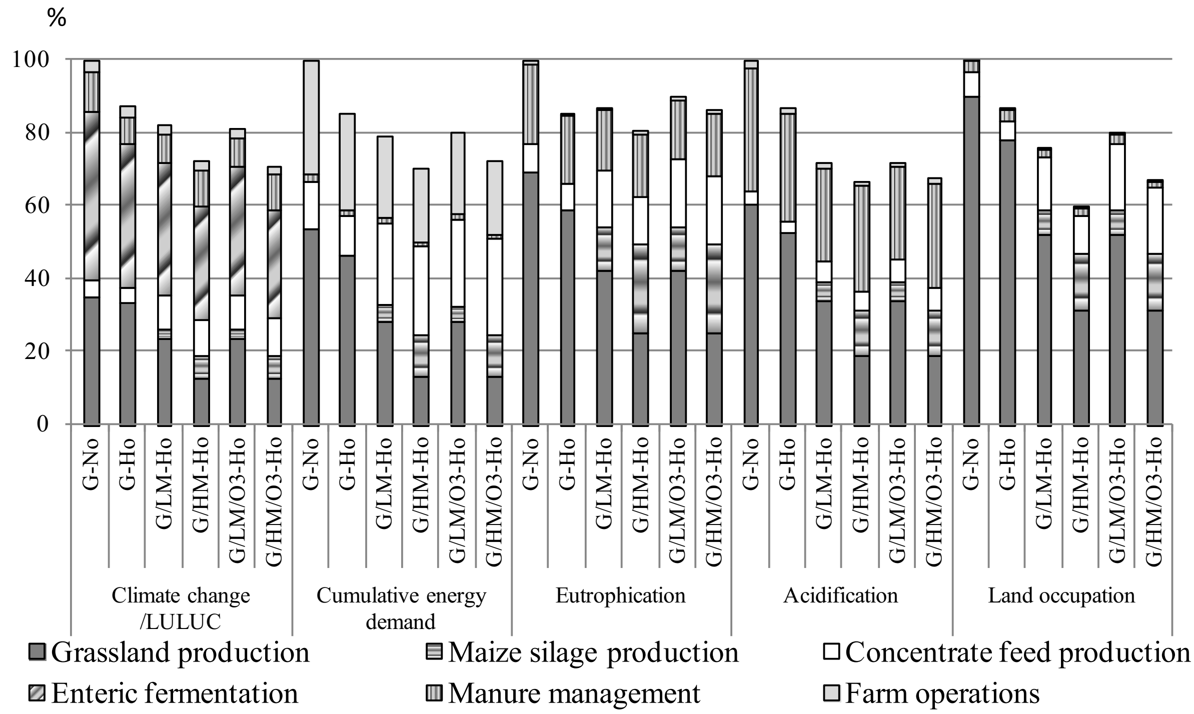

Figure 2. Contribution (in %) of main processes to environmental impacts of G-No, G-Ho, G/LM-Ho, G/HM-Ho, G/LM/O3-Ho and G/HM/O3-Ho dairy sub-systems[1] (G-No represents 100%; other systems are relative to G-No)

[1]Dairy sub-systems are defined in Materials and Methods
[2]Climate change including effect of land use and land-use change

Table 5. Environmental impacts of G-No, G-Ho, G/LM-Ho, G/HM-Ho, G/LM/O3-Ho and G/HM/O3-Ho dairy sub-systems[1] and whole farms (including dairy sub-system and cash-crop sub-system)

	G-No		G-Ho		G/LM-Ho		G/HM-Ho		G/LM/O3-Ho		G/HM/O3-Ho	
	Dairy sub-system	whole farm	Dairy sub-system	whole farm	Dairy sub-system	whole farm	Dairy sub-system	whole farm	Dairy sub-system	whole farm	Dairy sub-system	whole farm
Per dairy sub-system or whole farm												
Climate change, t CO_2 eq.	599	599	498	518	452	506	383	462	448	502	379	458
Climate change/LULUC[2], t CO_2 eq.	549	549	480	500	451	506	397	476	446	500	390	469
Cumulative energy demand, GJ eq.	1730	1730	1481	1584	1371	1666	1219	1655	1389	1685	1255	1692
Eutrophication, t PO_4^{3-}eq.	1.87	1.87	1.60	1.79	1.62	2.15	1.50	2.25	1.68	2.20	1.61	2.36
Acidification, t SO_2 eq.	4.74	4.74	4.12	4.24	3.40	3.72	3.17	3.61	3.42	3.73	3.21	3.65
Land occupation, ha*yr	61.3	61.3	52.9	59.7	46.4	64.4	36.3	62.0	48.8	66.8	40.9	66.5
Per ha of land occupation												
Climate change, t CO_2 eq.	9.8	9.8	9.4	8.7	9.7	7.9	10.6	7.5	9.2	7.5	9.3	6.9
Climate change/LULUC[2], t CO_2 eq.	9.0	9.0	9.1	8.4	9.7	7.9	10.9	7.7	9.1	7.5	9.5	7.1
Cumulative energy demand, GJ eq.	28.2	28.2	28.0	26.5	29.5	25.9	33.5	26.7	28.5	25.2	30.7	25.4
Eutrophication, kg PO_4^{3-}eq.	30.5	30.5	30.2	30.1	35.0	33.3	41.4	36.3	34.4	33.0	39.5	35.5
Acidification, kg SO_2 eq.	77.4	77.4	77.9	71.0	73.3	57.7	87.1	58.2	70.1	55.9	78.5	54.9
Land occupation, ha*yr	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0

[1]Dairy sub-systems are defined in Materials and Methods

[2]Climate change including effect of land use and land-use change

3.4. Dairy sub-system efficiency and impacts expressed per ha of land occupation

Per ha of land occupation, the G-Ho dairy sub-system used the same quantity of gross and net energy in feed but produced more nutritional energy and protein (13 and 5%, respectively) in animal products than the G-No dairy sub-system (Table 6). The conversion ratios from gross and net energy in feed to nutritional energy in animal products were 12% lower for the G-Ho than the G-No dairy sub-system. Per ha of land occupation, G/LM-Ho and G/HM-Ho dairy sub-systems produced more gross and net feed energy and more energy and protein in animal products than G-Ho. The conversion ratios from gross and net energy in feed to nutritional energy in animal products were, respectively, 7 and 4% lower for G/LM-Ho and 17 and 15% lower for G/HM-Ho than for the G-Ho dairy sub-system. Per ha of land occupation, G/LM/O3-Ho and G/HM/O3-Ho produced less energy in feed and less energy and protein in animal products than G/LM-Ho and G/HM-Ho dairy sub-systems, but energy conversion ratios were identical. The share of meat in the total animal outputs was lower for energy than for protein content.

Table 6. Efficiency of G-No, G-Ho, G/LM-Ho, G/HM-Ho, G/LM/O3-Ho and G/HM/O3-Ho dairy sub-systems[1] for the conversion from gross energy in feed to gross energy in animal products (expressed per ha of land occupation)

	G-No	G-Ho	G/LM-Ho	G/HM-Ho	G/LM/ O3-Ho	G/HM/ O3-Ho
Gross energy in feed (Mcal/ha)	32154	32127	33921	38233	32327	34378
Net energy in feed (Mcal/ha)	11117	11096	12124	13634	11531	12236
Gross energy in animal products (Mcal/ha)	3275	3700	4207	5319	4000	4729
Energy in milk (%)	95	96	96	97	96	97
Energy in meat (%)	5	4	4	3	4	3
Ratio gross feed energy/animal product energy	9.8	8.7	8.1	7.2	8.1	7.3
Ratio net feed energy/animal product energy	3.4	3.0	2.9	2.6	2.9	2.6
Total protein in animal products (kg/ha)	171	179	205	244	195	217
Protein in milk (%)	83	88	90	91	90	91
Protein in meat (%)	17	12	10	9	10	9

[1]Dairy sub-systems are defined in Materials and Methods

The impacts per ha of land occupation of the G-Ho dairy sub-system were 1-4% lower than those of G-No, except for CC/LULUC and AC (increase 1%) (Table 5). Among Holstein-cow-based dairy sub-systems, impacts per ha of land occupation were higher by 3-16% for G/LM-Ho and by 12-37% for G/HM-Ho compared to those of G-Ho, except for AC (only for

G/LM-Ho). Impacts per ha of land occupation of the G/LM/O3-Ho and G/HM/O3-Ho dairy sub-systems were 2-6% and 5-13% lower than those of G/LM-Ho and G/HM/O3-Ho, respectively.

4. Discussion

4.1. Impacts of milk: comparison with previous studies

A comparison with previous LCA studies of milk production was performed, according to grass-based or maize-silage-based dairy systems and to co-product handling methods (Table 7). With economic allocation, G/HM-Ho milk was compared to milk from a high maize-silage system in France (van der Werf et al., 2009) and a high N fertilisation and maize-silage system (NFMS) in New Zealand (NZ) (Basset-Mens et al., 2009), while G-Ho milk was compared to average NZ milk and high N fertilisation grass-based (NF) milk (Basset-Mens et al., 2009). With biophysical allocation, G-Ho milk was compared to Irish grass-based milk and G/LM-Ho to Irish confinement-system milk (O'Brien et al., 2012). Per kg of FPCM, our results for CC and AC were 23 and 43% higher, respectively, but for EP and LO were 23 and 13% lower, respectively, than those reported by van der Werf et al. (2009). Results in Basset-Mens et al. (2009) were reported per kg of milk (uncorrected milk), but for average NZ milk, 1 kg milk corresponded to 1.09 kg FPCM. Overall, our results for CC, AC, EP and LO were 44-129% higher than those reported by Basset-Mens et al. (2009). Our results for CC/LULUC, EP, AC and LO were 29-95% higher than those reported by O'Brien et al. (2012) for grass-based milk and 3-42% higher for confinement-system milk, except for AC (17% lower). Dairy systems in NZ are based exclusively on high-quality pasture with a long grazing season (pasture provides 92% of feed intake for the herd) which requires less N fertiliser (93, 108 and 81 kg N/ha/yr for average NZ, NF and NFMS, respectively) and has a high yield (10, 12 and 10 t DM/ha/yr for average NZ, NF and NFMS, respectively) (calculated from on-farm and off-farm pasture, Basset-Mens et al., 2009). In the grass-based dairy system in Ireland, grassland also had a high yield (12 t DM/ha/yr), but required a high amount of N fertiliser (260 kg N/ha/yr) (O'Brien et al., 2012). However, in the confinement-dairy system, grassland only required 85 kg N/ha/yr and yielded 9 t DM/ha/yr (O'Brien et al., 2012). In our study, grassland required in average of 171, 175 and 145 kg N/ha/yr and yielded an average of 9 t DM/ha/yr for G-Ho, G/LM-Ho and G/HM-Ho dairy sub-systems, respectively. Furthermore, the emission factor for N_2O from N deposited by animals during grazing was 1% in NZ dairy systems, based on IPCC-NZ methodology (Basset-Mens et al., 2009), compared to a default emission factor of 2% used in van der Werf et al. (2009), O'Brien et al. (2012) and the present study, based on IPCC methodology (IPCC, 2006). Differences can be partly explained by higher replacement rates (31-41%) in dairy systems of the present study than those in Basset-Mens et al. (2009) (22%) and O'Brien et al. (2012) (18%). Consequently, more heifers are needed to replace cull cows; that is, more inputs are used, and associated emissions of these heifers are added to the total impacts of the dairy sub-system and the milk it produces. In addition, cows had their first calving at 3 years in our study (for G-Ho and G/LM-Ho) compared to 2 years in G/HM-Ho (this study) and in the systems described by O'Brien et al. (2012). Differences may come from the use of different databases (e.g., older databases, such as BUWAL (1996) in van der Werf et al. (2009)) for impact assessment of associated upstream process.

Table 7. Reference studies, systems used, and potential impacts according to functional units and co-product handling methods compared to particular sub-systems in the present study

Reference studies	van der Werf et al. (2009)	Basset-Mens et al. (2009)			O'Brien et al. (2012)	
Systems used	Conventional	Average NZ	NF	NFMS	Grass	Confinement
Silage maize area in total forage area (%)	29.5	1.7	0	26.6	0	non available
Allocation method	Economic	Economic	Economic	Economic	Biophysical	Biophysical
Functional unit (FU)	kg FPCM	kg milk	kg milk	kg milk	kg FPCM	kg FPCM
Milk produced (FU/cow/year)	7678	3764	4718	4848	6538	7942
Environmental impacts						
Climate change (kgCO_2 eq.)	1.04	0.93	0.76	0.75		
Climate change/LULUC[1] (kgCO_2 eq.)					0.87	1.03
Energy use (MJ)	2.80	1.51	1.13	1.55	2.30	3.90
Eutrophication (g PO_4^{3-} eq.)	7.10	2.93	2.50	2.38	3.40	4.60
Acidification (g SO_2 eq.)	7.60	8.12	6.74	5.78	6.90	11.90
Land competition (m^2*yr)	1.37	1.15	0.80	0.72	0.73	0.93
Compared with dairy sub-system[1]	G/HM-Ho	G-Ho	G-Ho	G/HM-Ho	G-Ho	G/LM-Ho

[1]Climate change including effect of land use and land-use change

Some of the differences with other studies can be explained by the way in which co-product handling methods were applied. We subdivided unit processes (e.g., cash crops and animal products), as argued by van der Werf et al. (2009), to avoid attributing environmental interventions associated with animals (e.g. CH_4 emissions) to crop products and vice-versa. For CC impacts, most studies did not include C emissions due to LUC and C sequestration in permanent grassland (Basset-Mens et al., 2009; van der Werf et al., 2009; O'Brien et al., 2012). Comparing LCAs of milk production is difficult due to lack of detail in previous studies and differences in assumptions, system boundaries and methods used for calculating inventories (Basset-Mens et al., 2009; van der Werf et al., 2009).

4.2. Effect of rations and production systems

Differences in environmental impacts between dairy sub-systems in the present study were related to differences in management strategies (feeding strategy, herd management and grassland management). Firstly, lactation performance depends on feeding strategies (Delaby et al., 2009), including type of ration and total amount of net energy intake, as illustrated by differences between G/LM-Ho and G-Ho dairy sub-systems. Annual milk yield per cow was higher (by 1.43 t FPCM) in G/LM-Ho than in G-Ho due to (1) the ration (based in the former on maize silage and concentrate (70:30) in winter and concentrate supplements in the grazing period and in the latter on grass silage and concentrate (85:15) in winter and no supplements in the grazing period) and (2) higher (27%) total net energy intake per cow in G/LM-Ho than in G-Ho. In consequence, G-Ho needed more cows, and thus more replacement heifers, to produce the same amount of milk as G/LM-Ho, even though cows in these two systems had the same reproductive performance (i.e. replacement rate) and age at first calving. Management strategies applied for heifers from birth to their first calving and for finishing cull cows were identical. Impacts related to each process (e.g. total feed production, enteric fermentation, manure management, farm operations) in G-Ho were higher than those of G/LM-Ho (except for total feed production for EP). Furthermore, impacts related to feed production expressed per net energy intake unit for G/LM-Ho were 2-16% lower than those for G-Ho, except for CED and EP (increase 1 and 11%, respectively) (results not shown). In this case, the use of maize silage and concentrate to increase milk yield per cow decreased impacts (except EP) of milk and meat from the dairy herd compared to those from a grass-based feeding strategy.

Compared to G/LM-Ho, milk production in G/HM-Ho was higher due to the larger proportion of maize silage in the total annual diet (52%, vs. 25% in G/LM-Ho) and the higher total net energy intake per cow (10% higher than in G/LM-Ho). Also, to ensure high milk production (i.e. high intake), calving was scheduled at the end of summer; therefore, G/HM-Ho had a shorter grazing period than other dairy sub-systems and supplemented maize silage during grazing. Consequently, G/HM-Ho could produce the same amount of milk as G/LM-Ho and G-Ho with a smaller herd. Another factor that decreased impacts of G/HM-Ho was cows having their first calving at 2 years, which strongly decreased impacts of replacement heifers. Furthermore, silage maize required less mineral fertiliser and had a higher yield per ha than grassland.

The use of maize silage in cow diets leads to increasing the use of soybean meal to satisfy crude protein requirements, as maize silage has less protein than grass grazed and grass silage. This may increase CO_2 emissions due to the use of soybeans produced on recently deforested areas. Feed-production impacts expressed per unit of net energy intake for G/HM-Ho were 7-16% lower than those for G/LM-Ho, except for CED (similar) and EP (increase 1%) (results not shown). Enteric methane produced by cows was lower with a ration based on maize silage than with one based on grass (Martin et al., 2010). In this case, feeding strategy along with other management strategies reduced impacts (except EP) of milk and meat from dairy production. Greater efficiency of animal production in maize-silage-based dairy sub-systems than grass-based dairy sub-systems was also illustrated by the former's lower conversion ratios of gross and net energy in feed to gross energy in animal products. Nevertheless, grass-based dairy sub-systems had a higher proportion of meat in total animal-protein outputs, especially G-No. Dollé et al. (2011) and Van Middelaar et al. (2011) also observed lower CC impact of milk from maize-silage-based dairy systems than grass-based dairy systems. However, it became higher in the former than in the latter when including C sequestration by grassland and inter-field hedgerows (Dollé et al., 2011) or C emissions due to ploughing of grassland for silage maize (Van Middelaar et al., 2011).

Including LULUC emissions is the subject of much debate, due to a lack of consensus on methodology (Henriksson et al., 2011) and the lack of a harmonised source of data. Our estimate of C sequestration for permanent grassland is based on data for C sequestration in French agricultural soils and is lower than those reported by Soussana et al. (2010) for European conditions. Accounting for grassland C sequestration and LULUC decreased the climate change impact (i.e. CC/LULUC vs. CC) of grassland production in each dairy sub-system by 12-13%; however, that of other feed production increased (by 50% for G-No, G-Ho, and G/LM-Ho; 72% for G/HM-Ho; 42% for G/LM/O3-Ho and 64% for G/HM/O3-Ho) due to the use of soybean meal (Fig. 2). Overall, net CC impacts slightly decreased in G-No and G-Ho (by 3 and 4%, respectively) but slightly increased in G/HM-Ho and G/HM/O3-Ho (by 4 and 3%, respectively). Thus effects of grassland C sequestration were compensated by emissions due to the use of soybean in G/LM-Ho and G/LM/O3-Ho. It should be noted that this study did not investigate the effect of farm conversion, i.e. the conversion from a grass-based system to a maize-silage-based system and consequently the ploughing of permanent grassland for annual cropping. Clearly ploughing permanent grassland for any reason will increase GHG emissions of dairy farms (Van Middelaar et al., 2011; Vellinga and Hoving, 2011).

Increasing the proportion of silage maize in the forage area increased nitrate leaching (Vertès et al., 2012). This explained why EP of milk and meat from G/LM-Ho and G/HM-Ho was higher than those from G-Ho. In fact, nitrate leaching is higher for silage maize than for other annual crops and lowest for grassland. It is especially high for silage maize or annual crops planted after temporary grassland (Vertès et al., 2012). It should further be noted that impacts on soil quality and macro-scale biodiversity were not included in this study due to the lack of appropriate methodology. However, silage maize is much more vulnerable to soil erosion than grassland and may pose a much larger threat to soil quality than grassland on erosion-prone soils. Similarly, grassland tends to have higher macro-scale biodiversity, than maize (e.g., Robertson et al., 2012). So, a grass-based dairy sub-system likely has lower impacts on soil quality and biodiversity than one with a major proportion of silage maize.

Intensification of dairy production, i.e. increasing milk production per cow, also requires using more concentrated feeds, principally based on cereals, which increases competition with human food. Nevertheless, this study shows that intensifying dairy production by feeding more maize silage and concentrated feed reduces the on-farm and off-farm area needed to produce the same amount of milk. As arable land is a limited resource, land no longer needed for milk production can produce crops for food or bioenergy. It has been suggested that improving cow productivity may be a viable GHG-mitigation approach for dairy systems (Gerber et al., 2011, Vellinga et al., 2011); this is confirmed by the lower GHG emissions per kg of milk and meat for the intensive systems in this study. However, this study also revealed that intensifying dairy production increased eutrophication per unit area, which may have major consequences for the local environment, in particular in regions that combine high animal densities and vulnerable ecosystems.

4.3. Effect of cow breed

The use of the dual-purpose breed (Normande) in the G-No dairy sub-system required more cows to produce the same amount of milk as the specialised milk breed (Holstein) in the G-Ho dairy sub-system. Impacts both per kg FPCM and per dairy sub-system were higher for G-No than for G-Ho regardless of impact category (except EP) and co-product handling method. Estimates of CC impact agree with those of Zehetmeier et al. (2012), who compared GHG emissions of three systems producing the same amount of milk, based either on a dual-purpose breed (Fleckvieh) yielding 6000 or 8000 kg milk/cow/yr or Holstein yielding 10000 kg milk/cow/yr. The Fleckvieh dairy sub-systems, however, produced more meat than the Holstein dairy sub-system. Zehetmeier et al. (2012) showed that if milk and meat production were kept constant (by replacing decreased beef production with beef from suckler cows via system expansion), GHG emissions of the Holstein system were 8% higher than those of the two Fleckvieh systems. Martin and Seeland (1999) also concluded that the specialisation in both milk and beef production marginally increases emissions per kg of total protein produced, if beef production were to be completely balanced. In our study, impacts of Holstein sub-systems with suckler-beef-cattle production included to produce the same amount of milk and meat as the Normande sub-system were lower than those of the Normande sub-system by 8-15% for CC, 8-11% for CC/LULUC, 7-9% for CED and 7-17% for AC. As for other impacts, there was a 4% increase in EP for G/LM-Ho and G/HM-Ho and an increase in LO by 1% for G-Ho and 5% for G/LM-Ho. Thus, overall, a Holstein dairy sub-system combined with suckler beef cattle had lower environmental impacts than a Normande dairy sub-system when producing the same amount of milk and meat. Our results seem to contradict those reported by Zehetmeier et al. (2012); however, the two studies are not comparable, as Holstein cows in our dairy sub-systems yielded 6.7-8.7 t FPCM/cow/yr, which was in the range of those of the Fleckvieh (6 and 8 t/cow/yr) but much lower than that of the Holstein in Zehetmeier et al. (2012) (10 t/cow/yr). The higher the milk production, the lower the meat-to-milk ratio; consequently, a larger amount of meat from suckler beef cattle is required, explaining the higher GHG emissions of the Holstein than the Fleckvieh in Zehetmeier et al. (2012).

In the present study, the on-farm UAA of Holstein farms which was not used for milk production (due to higher productivity of the dairy sub-system) was used to produce wheat and rapeseed cash crops. The impacts of Holstein farms (dairy plus cash-crop sub-systems) were lower than those of the Normande farm (only a dairy sub-system). Therefore, if demand for meat decreases, the use of highly productive dairy cows can both increase cereal production and decrease environmental impacts of the cattle production sector. However, the effects of increasing dairy cow productivity on environmental impacts of the entire dairy and beef production sector at a regional or global scale depend on the demand for milk and meat and the ratio of milk and meat produced per dairy cow (Zehetmeier et al., 2012).

4.4. Effect of enriching diets with omega-3 FAs

Grass and grass-based forages (grass silage and bale silage) have higher omega-3 FA content than maize silage (Ferlay et al., 2008) and rations based on grass during grazing (for all dairy sub-systems) and grass silage in winter (for G-No and G-Ho) were sufficient in omega-3 FA supply for cows. Increasing the omega-3 FA content in cow diets, however, slightly increases omega-3 FA content in milk and meat from dairy production (Doreau et al., 2011). Therefore, extruded linseed was only added to increase the FA content of cow diets in winter and during transition periods for G/LM-Ho and G/HM-Ho dairy sub-systems. The use of extruded linseed decreased enteric CH_4 emissions by 3.5 and 6.0% for G/LM/O3-Ho and G/HM/O3-Ho dairy sub-systems compared to G/LM-Ho and G/HM-Ho, respectively. Nevertheless, CC/LULUC impacts of concentrate feed for G/LM/O3-Ho and G/HM/O3-Ho were higher than those of G/LM-Ho and G/HM-Ho (by 2.4 and 5.0%, respectively), even though the use of extruded linseed (with high energy and protein contents) reduced the use of wheat and soybean meal. Therefore, in terms of CC impact, the decrease in enteric CH_4 emissions due to including extruded linseed in diets was nearly offset (decreased 1.2 and 1.8% for G/LM/O3-Ho and G/HM/O3-Ho, respectively) by the GHG emissions from producing the concentrated feed containing it. The inclusion of extruded linseed also increased the EP and LO impacts of the dairy sub-system.

4.5. Effect of co-product handling methods

The method used to handle co-products has a strong influence on estimated environmental impacts of milk and animals sold for meat from dairy production systems (FAO, 2010; Kristensen et al., 2011; Flysjö et al., 2011). In an LCA study, if subdivision of processes is not possible, ISO standards (ISO, 2006) first recommend system expansion to deal with co-products. For dairy systems, boundaries can be "expanded" to include alternative ways to produce the co-products of milk production (i.e. beef from surplus calves and cull cows). Thomassen et al. (2008) and Kristensen et al. (2011) assumed that dairy-system beef could be replaced by a mix of suckler-cow beef and pork, while Cederberg and Stadig (2003) used beef from a suckler-cow system. Flysjö et al. (2011) tested expansion scenarios that replaced dairy-system beef with suckler-cow beef; a mix of suckler-cow beef and pork; or a mix of suckler-cow beef, pork and chicken. In the present study, we assumed that beef from dairy-system was replaced by that from suckler-cow system, as they have similar quality and price, and France has the highest annual per-capita beef consumption among European countries (FranceAgriMer, 2010). However, the impacts per kg of meat from dairy systems (depending on co-product handling methods) were generally lower than those from suckler beef-cattle systems (as calculated from Kristensen et al. (2011), Zehetmeier et al. (2012) and the present study). As a result, when system expansion is used, the percentage of dairy sub-system environmental impacts attributed to its sold animals depends on the type of animals "used" as replacement (Flysjö et al., 2011) and on these animals' impacts. For example, sold animals received 22-25% of dairy sub-system impacts in Kristensen et al. (2011) (CC only); 13-40% in Cederberg and Stadig (2003) for CC, AC, EP, LO and energy use and; in the present study, 27-51% for CC, CC/LULUC, AC, EP, CED and 69-85% for LO.

In other studies, as in our study, impacts per kg of milk were usually lower with system expansion than with allocation methods (Cederberg and Stadig, 2003; Kristensen et al., 2011; Flysjö et al., 2011). If allocation is unavoidable, ISO standards prefer allocation based on (in descending order) biophysical causality, energy/protein content or economic value. When applying the biophysical allocation method developed by IDF (2010) in the present study, the percentage of impacts of sold cows and replacement heifers varied from 17-28% (depending on the system), similar to results of Kristensen et al. (2011) (24-29%) but higher than those of Cederberg and Stadig (2003) and Basset-Mens et al. (2009), who used a allocation factor of 15% for sold animals according to energy and protein requirements to cover dairy-cow milk production, maintenance and pregnancy. In the present study, the percentages of dairy sub-system impacts attributed to sold animals with protein or economic allocation were similar to those of Kristensen et al. (2011). Protein allocation was recommended by the FAO (2010) because it reflects a primary function of dairy production (to provide humans with animal protein) and does not depend on time and place like economic allocation. Nonetheless, economic allocation is the most common method used for co-product handling (de Vries and de Boer, 2009), reflecting that products tend to be produced for their economic value (Jolliet et al., 2010).

5. Conclusion

This study demonstrated that grass-based dairy production contributed more to climate change, energy demand and land occupation (both per kg of milk or per dairy sub-system) than maize-silage-based dairy production. However, from a local or territorial point of view, it should be noted that grass-based dairy production had lower impacts per ha of land occupied, in particular for eutrophication. The enrichment of rations with omega-3 fatty acids to reduce enteric methane slightly reduced climate change impact, but did not reduce the impacts of the whole production system. Dairy systems using Holstein, a specialised milking breed, had lower impacts than the system using Normande, a dual-purpose breed. The methodological choice for co-product handling methods largely affected the impacts of milk and animal outputs. This study compared scenarios for dairy-production systems according to an attributional LCA approach. Its results, in particular the comparison of grass-based versus silage-maize-based milk production, should not be used to identify impact-mitigation options, in particular the partial replacement of grassland by silage maize to reduce GHG emissions. Further studies focused on mitigation strategies should include direct and indirect consequences of implementation of mitigation strategies for different environmental impacts.

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Supplementary materials

Table S1. Annual inputs used, dry matter yield[1], nitrate-N emitted and pasture residence time[2] for grassland used for cutting and/or grazing in different dairy sub-systems[3] (G-No, G-Ho, G/LM-Ho, G/LM/O3-Ho, G/HM-Ho, G/HM/O3-Ho)

	Area ha	**N** mineral kg/ha	**N** manure kg/ha	**N** deposited by grazing kg/ha	**Seed** kg/ha	**Pesticide** (active ingredient) kg/ha	**Diesel** kg/ha	**Agricultural machinery** kg/ha	**Plastic for silage** kg/ha	**Conserved forage yield** t/ha	**Grazing yield** t/ha	**Nitrate-N emitted** kg/ha	**Pasture residence time** LU days/ha
G-No													
Temporary PRG-C[4] for cutting and heifer grazing	2.5	60	105	150	5	0.09	48	15	6	4	5.4	15.0	250
Temporary PRG-C for cutting and cow grazing	7.2	60	105	88	5	0.09	88	22	17	5.8	3.6	15.0	183
Temporary PRG[5] for cutting and cow grazing	2.1	200	120	88	5	0.09	91	21	6	5.8	3.6	15.0	183
Permanent grassland for heifer grazing	13.1	150	0	208	0	0.09	24	3	0	0	7.5	37.0	480
Permanent grassland for cow grazing	12.3	250	0	250	0	0.09	24	3	0	0	10.2	44.5	542
Temporary PRG for cutting	11.0	200	105	0	5	0.09	149	37	8	8.9	0	5.0	0
PRG ley for cutting	4.0	200	120	0	25	0.09	179	43	8	8.9	0	5.0	0
Permanent grassland for cutting	2.8	200	105	0	0	0.09	142	35	8	8.9	0	5.0	0
G-Ho													
Temporary PRG-C for cutting and heifer grazing	2.0	60	105	150	5	0.09	48	15	6	4	5.4	15.0	250
Temporary PRG-C for cutting and cow grazing	7.8	60	105	85	5	0.09	88	22	17	5.8	3.6	15.0	183
Permanent grassland for heifer grazing	10.8	150	0	208	0	0.09	24	3	0	0	7.5	37.0	480
Permanent grassland for cow grazing	10.3	250	0	240	0	0.09	24	3	0	0	10.2	44.5	542
Temporary PRG for cutting	9.8	200	105	0	5	0.09	149	37	8	8.9	0	5.0	0
PRG ley for cutting	3.5	200	120	0	25	0.09	179	43	8	8.9	0	5.0	0
Permanent grassland for cut	3.5	200	105	0	0	0.09	142	35	8	8.9	0	5.0	0
G/LM-Ho and G/LM/O3-Ho													
Temporary PRG-C for cutting and heifer grazing	1.7	60	105	156	5	0.09	48	15	6	4	5.4	15.0	250
Temporary PRG-C mixture for cutting and cow grazing	3.8	60	105	82	5	0.09	84	21	6	5.8	3.6	15.0	183
Temporary PRG for cutting and cow grazing	2.7	200	120	82	5	0.09	91	21	6	5.8	3.6	15.0	183

Permanent grassland for heifer grazing	9.0	150	0	217	0	0.09	24	3	0	0	7.5	37.0	480
Permanent grassland for cow grazing	8.5	250	0	233	0	0.09	24	3	0	0	10.2	44.5	542
Temporary PRG for cutting	4.0	200	105	0	5	0.09	149	37	8	8.9	0	5.0	0
PRG ley for cutting	2.1	200	120	0	25	0.09	179	43	8	8.9	0	5.0	0
G/HM-Ho and G/HM/O3-Ho													
Temporary PRG-C mixture for cutting and heifer grazing	3.7	60	105	136	5	0.09	47	14	6	4	5.4	15.0	250
Temporary PRG for cutting and cow grazing	2.3	130	120	89	5	0.09	51	13	6	4	4.4	15.0	292
Temporary PRG for heifer grazing	1.8	130	0	211	5	0.09	25	4	0	0	8.4	37.0	480
Permanent grassland for heifer grazing	2.0	150	0	188	0	0.09	19	3	0	0	7.5	37.0	480
Permanent grassland for cow grazing	8.0	180	0	186	0	0.09	19	3	0	0	9.2	38.0	489
Temporary PRG for cutting	1.5	200	105	0	5	0.09	149	37	8	8.9	0	5.0	0

[1]Yield of grassland corresponds to the yield obtained when all grass is machine harvested. Conserved forage consisted of hay, grass silage and/or bale silage. Losses during conservation and feeding were assumed to be 15% of the initial DM. Grass not harvested as conserved forage was available for intake by animals during grazing. For several reasons (selective grazing, trampling of grass, unfavourable weather conditions) a part of the grass grown is not ingested, this "loss" corresponded to 15% of grass dry matter available for grazing.

[2]Estimated as 20.4 h per day of "full pasture" per Livestock Unit (LU) and 8.4 h per day of "transition to pasture" per LU, summed and converted to LU days

[3]Dairy sub-systems are defined in Materials and Methods

[4]Perennial ryegrass-clover mixture

[5]Perennial ryegrass

Table S2. Environmental impacts per ha of grassland production in G-No, G-Ho, G/LM-Ho,

	Climate change, t CO_2 eq.	Climate change /LULUC[2], t CO_2 eq.	Cumulative energy demand, GJ eq.	Eutrophication, kg PO_4^{3-}eq.	Acidification, kg SO_2 eq.
G-No					
Temporary PRG-C[3] for cutting and heifer grazing	3.31	1.66	8.15	23.41	69.68
Temporary PRG-C for cutting and cow grazing	2.94	1.29	12.00	21.60	61.61
Temporary PRG[4] for cutting and cow grazing	4.62	2.97	22.18	16.40	36.87
Permanent grassland for heifer grazing	4.37	3.63	10.44	27.10	42.70
Permanent grassland for cow grazing	6.25	5.52	16.34	33.73	56.85
Temporary PRG for cutting	4.18	2.53	24.12	16.60	58.01
PRG ley for cutting	4.78	4.78	26.74	10.95	31.61
Permanent grassland for cutting	4.05	3.32	23.31	16.51	57.69
G-Ho					
Temporary PRG-C for cutting and heifer grazing	3.31	2.95	8.15	23.40	69.66
Temporary PRG-C for cutting and cow grazing	2.88	2.52	12.00	21.40	60.74
Permanent grassland for heifer grazing	4.36	3.63	10.44	27.10	42.68
Permanent grassland for cow grazing	6.11	5.37	16.34	33.20	54.39
Temporary PRG for cutting	4.18	3.81	24.12	16.60	58.01
PRG ley for cutting	4.78	4.78	26.74	10.95	31.61
Permanent grassland for cut	4.05	3.32	23.31	16.51	57.69
G/LM-Ho and G/LM/O3-Ho					
Temporary PRG-C for cutting and heifer grazing	3.32	2.95	8.15	23.41	69.69
Temporary PRG-C for cutting and cow grazing	2.86	2.50	10.76	21.45	60.87
Temporary PRG for cutting and cow grazing	4.89	4.52	19.38	17.71	42.60
Permanent grassland for heifer grazing	4.37	3.63	10.44	27.11	42.72
Permanent grassland for cow grazing	6.18	5.44	16.34	33.42	55.34
Temporary PRG for cutting	4.18	3.81	24.12	16.60	58.01
PRG ley for cutting	4.78	4.78	26.74	10.95	31.61
G/HM-Ho and G/HM/O3-Ho					
Temporary PRG-C for cutting and heifer grazing	3.16	2.79	8.01	22.79	67.09
Temporary PRG for cutting and cow grazing	3.60	3.24	15.09	15.50	33.21
Temporary PRG for heifer grazing	4.20	3.83	9.62	26.81	41.49
Permanent grassland for heifer grazing	4.14	3.40	10.12	26.25	39.05
Permanent grassland for cow grazing	4.70	3.96	11.90	27.66	43.31
Temporary PRG for cutting	4.18	3.81	24.12	16.60	58.01

G/LM/O3-Ho, G/HM-Ho, G/HM/O3-Ho dairy sub-systems[1]

[1]Dairy sub-systems are defined in Materials and Methods

[2]Climate change including effect of land use and land-use change

[3]Perennial ryegrass-clover mixture

[4]Perennial ryegrass

Table S3. Environmental impacts per t dry matter of ingested forages and other feed ingredients used and/or produced in G-No, G-Ho, G/LM-Ho, G/LM/O3-Ho, G/HM-Ho, G/HM/O3-Ho dairy farms[1]

Type of forage	Dairy farm	Climate change, kg CO_2 eq.	Climate change /LULUC[2], kg CO_2 eq.	Cumulative energy demand, MJ eq.	Eutrophication, kg PO_4^{3}-eq.	Acidification, kg SO_2 eq.	Land occupation, m^2*yr
Maize silage[3]	G/LM-Ho and G/LM/O3-Ho	240	240	1327	4.0	4.3	749
Maize silage[3]	G/HM-Ho and G/HM/O3-Ho	284	284	1532	3.6	4.5	750
Wheat	All	414	414	2191	4.1	2.5	1438
Rapeseed	G/LM-Ho, G/LM/O3-Ho, G/HM-Ho, G/HM/O3-Ho	1084	1084	6417	9.3	5.1	3119
Soybean meal	All	707	1544	11818	6.1	6.6	1644
Extruded linseed	G/LM/O3-Ho, G/HM/O3-Ho	1079	1079	9587	15.1	8.5	5735
Minerals	All	659	659	10301	4.7	9.3	93

[1]Dairy sub-systems are defined in Materials and Methods
[2]Climate change including effect of land use and land-use change
[3]Losses during conservation and feeding of maize silage were assumed to be 10% of the initial DM.

Table S4. Environmental impacts per kg live weight of pregnant heifer from G-No, G-Ho, G/LM-Ho, G/LM/O3-Ho, G/HM-Ho, G/HM/O3-Ho dairy sub-systems[1] according to four methods for co-product handling

	G-No	G-Ho	G/LM-Ho	G/HM-Ho	G/LM/O3-Ho	G/HM/O3-Ho
Biophysical allocation						
Climate change, kg CO_2 eq.	15.8	16.2	16.2	13.0	16.1	13.0
Climate change/LULUC[2], kg CO_2 eq.	14.9	15.5	15.5	12.9	15.5	12.9
Cumulative energy demand, MJ eq.	37.8	38.7	38.5	27.8	38.6	27.9
Eutrophication, g PO_4^{3-}eq.	43.5	44.6	44.7	40.3	44.8	40.6
Acidification, g SO_2 eq.	93.6	95.9	94.6	82.7	94.6	82.8
Land occupation, m^2*yr	17.9	18.4	18.3	15.0	18.4	15.1
Protein allocation						
Climate change, kg CO_2 eq.	15.6	16.1	16.0	12.9	16.0	12.9
Climate change/LULUC, kg CO_2 eq.	14.7	15.4	15.3	12.8	15.3	12.7
Cumulative energy demand, MJ eq.	37.2	38.2	38.1	27.5	38.1	27.5
Eutrophication, g PO_4^{3-}eq.	42.8	44.0	44.1	39.9	44.2	40.1
Acidification, g SO_2 eq.	91.9	94.4	93.4	81.8	93.4	81.9
Land occupation, m^2*yr	17.7	18.2	18.2	14.9	18.2	15.0
Economic allocation						
Climate change, kg CO_2 eq.	16.8	16.7	16.6	13.5	16.6	13.5
Climate change/LULUC, kg CO_2 eq.	15.8	16.0	15.9	13.4	15.9	13.3
Cumulative energy demand, MJ eq.	40.7	40.1	39.9	29.3	39.9	29.5
Eutrophication, g PO_4^{3-}eq.	46.6	46.2	46.3	42.2	46.5	42.6
Acidification, g SO_2 eq.	101.7	99.8	98.0	86.7	98.0	86.8
Land occupation, m^2*yr	18.9	18.9	18.8	15.4	18.8	15.6
System expansion						
Climate change, kg CO_2 eq.	16.4	16.9	16.9	14.1	16.9	14.1
Climate change/LULUC, kg CO_2 eq.	15.4	16.0	16.0	13.6	16.0	13.7
Cumulative energy demand, MJ eq.	38.3	39.4	39.4	29.2	39.6	29.4
Eutrophication, g PO_4^{3-}eq.	45.8	47.1	47.1	43.6	47.1	43.5
Acidification, g SO_2 eq.	94.6	97.4	97.4	86.6	97.6	86.9
Land occupation, m^2*yr	19.8	20.4	20.4	17.7	20.3	17.7

[1]Dairy sub-systems are defined in Materials and Methods

[2]Climate change including effect of land use and land-use change

Chapter 5

Consequential LCA of switching from maize-silage-based to grass-based dairy systems

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Consequential LCA of switching from maize-silage-based to grass-based dairy systems

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Abstract

Purpose: This study aimed to investigate the environmental consequences (on climate change and land use) of an increase in preference for grass-based milk in France using a consequential life cycle assessment (CLCA) approach. This increase in preference was assumed to be satisfied domestically, by converting maize-silage-based dairy farms (MS farm) to grass-based dairy farms (G farm) while keeping on-farm usable agricultural area and total milk production of farm constant.

Methods: The possible consequences of an increase in preference for grass-based milk were identified based on cause-and-effect relationships. The conversion from MS to G reduced the use of soybean meal, changed the on-farm cropping pattern, and produced more animals but less wheat and no rapeseed. Effects on on-farm soil C were predicted with the RothC model and on global land use change (LUC) with models of global agricultural markets (GTAP and LEITAP). System expansion using animals from a suckler-beef production system was applied to estimate the impacts of milk and animal co-products from the dairy system. Land occupation and climate change impacts were estimated. The consequences of farm conversion were attributed only to the milk, as preference for grass-based milk drove the conversion process.

Results and discussion: The conversion from MS to G increases land occupation and climate change impact for the G farm, respectively, by 9 and 7% according to GTAP and 14 and 51% according to LEITAP. Land occupation and climate change impact of milk produced by the G farm after conversion increased, respectively, by 82 and 13% with GTAP and 123 and 97% with LEITAP relative to those for the MS farm (before conversion). The production of additional wheat and rapeseed outside the G farm increased impacts of the G farm (by 29-69% depending on impacts and model used). Results indicate that the farm conversion would have substantial consequences on global LUC and that it is important to account for this in a LCA approach.

Conclusions: This study demonstrated how environmental consequences of an increased preference for grass-based milk can be estimated with CLCA. Land-use and land-use change (LULUC) contributed substantially to the impacts of grass-based milk, and results were highly sensitive to the LULUC model used. The many possible chain-of-event pathways that follow a change in preference for a given product yield high uncertainty in CLCA results.

Recommendations: This study only assessed one possible way to meet the increase in preference for grass-based milk; it is necessary to perform a sensitivity analysis to investigate other possible scenarios resulting from this increase in preference. Further studies are required to integrate global economic models in CLCA modelling to identify and assess the processes affected by a change in preference.

Keywords: Consequential LCA, grass-based milk, maize-silage-based milk, LULUC, climate change, land occupation

1. Introduction

Worldwide, the dairy sector is estimated to contribute 4% to total global anthropogenic greenhouse gas (GHG) emissions (FAO, 2010). The dairy sector is estimated to contribute 20-30% to total GHG emissions of European Union (EU) livestock production (Weiss and Leip, 2012). In France, 50% of milk quota (representing 46% of dairy farms) comes from production systems with a high percentage (>30%) of silage maize in the total forage area (Institut de l'Elevage, 2009). In contrast, grass-based milk production (i.e. <5% of silage maize and > 80% of grassland in the total forage area) concerns only 11% of dairy farms and 8% of milk quota (Institut de l'Elevage, 2009). Grass-based milk production reduces feed cost and farm sensitivity to increased input prices and economic fluctuations (Institut de l'Elevage, 2009). Compared to silage maize, grassland also increases C sequestration in soil (Soussana et al., 2010), reduces nitrate leaching (Vertès et al., 2012) and enhances macro-scale biodiversity of the production system (e.g., Robertson et al., 2012). Grass-based milk has higher nutritional quality than maize-silage-based milk due to higher linolenic acid content in milk fat (Chilliard et al., 2001). Thus grass-based dairy production systems appear to have several economic and environmental advantages, and development of these systems has been encouraged in France over the past decade (Alard et al. 2002, Peyraud et al., 2009).

The Consequential Life Cycle Assessment (CLCA) approach aims to describe how physical flows may change in response to possible changes in the life cycle of the product under study (Ekvall and Weidema, 2004). This study investigated the potential consequences on climate change and land use of an increase in preference for grass-based milk in France using the CLCA approach. We assumed that a number of French consumers would begin purchasing grass-based (G) milk instead of "conventional" milk (i.e., maize-silage-based (MS) milk). We then assumed that this change in preference would be met exclusively by modifying domestic production, that is, by converting French dairy farms (represented by a single farm in this study) from a feeding system based to a large extent on silage maize to one based mainly on grass. As a G farm occupies more land than a MS farm to produce the same amount of milk (Delaby and Pavie, 2008), we assumed that the representative G farm would be configured to have the same usable agricultural area (UAA) and milk production as it did when it was an MS farm.

2. Materials and methods

2.1. Conversion of dairy farm

2.1.1. Maize-silage-based dairy farm

The MS farm occupies 55 ha of UAA, including a dairy production subsystem (28.7 ha) with a quota of 250 000 l milk and a cash-crop subsystem (26.3 ha). This farm corresponds to the HM-H dairy farm described in Chapter 4. The cash-crop subsystem produces wheat (17.8 ha) and rapeseed (8.5 ha). The MS dairy subsystem has highly productive Holstein cows (8.66 t Fat and Protein Corrected Milk (FPCM)/cow/yr) and 33% of its forage area occupied by silage maize. The herd consists of 32 cows (11 primiparous) that annually provide 29 calves, 13 of which are kept to be raised as heifers to replace cull cows. During the indoor period (September to mid-March), cows are fed mainly maize silage and concentrate feed. During the grazing period (120 days, including transition), cows mainly graze but are supplemented with maize silage and concentrate feed. Cows have their first calving at 25 months, and all calving is grouped at the end of summer. During drying-off, cows graze and are supplemented with maize silage and concentrate feed.

2.1.2. Grass-based dairy farm

The G farm, converted from the MS farm on the same 55 ha of UAA (Fig. 1), corresponds to the G-H dairy farm described in Chapter 4. The number of cows increases, as cows in the G farm yield less milk per year (6.74 t FPCM) than those in the MS farm. Thus, the herd consists of 41 cows (17 primiparous) that annually provide 38 calves, 19 of which are raised as heifers to replace cull cows. During the indoor period (mid-December to March), cows are fed mainly conserved grass and concentrate feed. During the grazing period (205 days, including transition), cows mainly graze and are supplemented with grass silage and minerals. Cows have their first calving at 36 months, as heifers have a low growth rate due to their grass-based diets, and all calving is grouped at the end of autumn. Drying-off occurs during the indoor period, with cows fed grass silage. Total forage area for this herd is 47.6 ha, leaving 7.4 ha for wheat in the cash-crop subsystem.

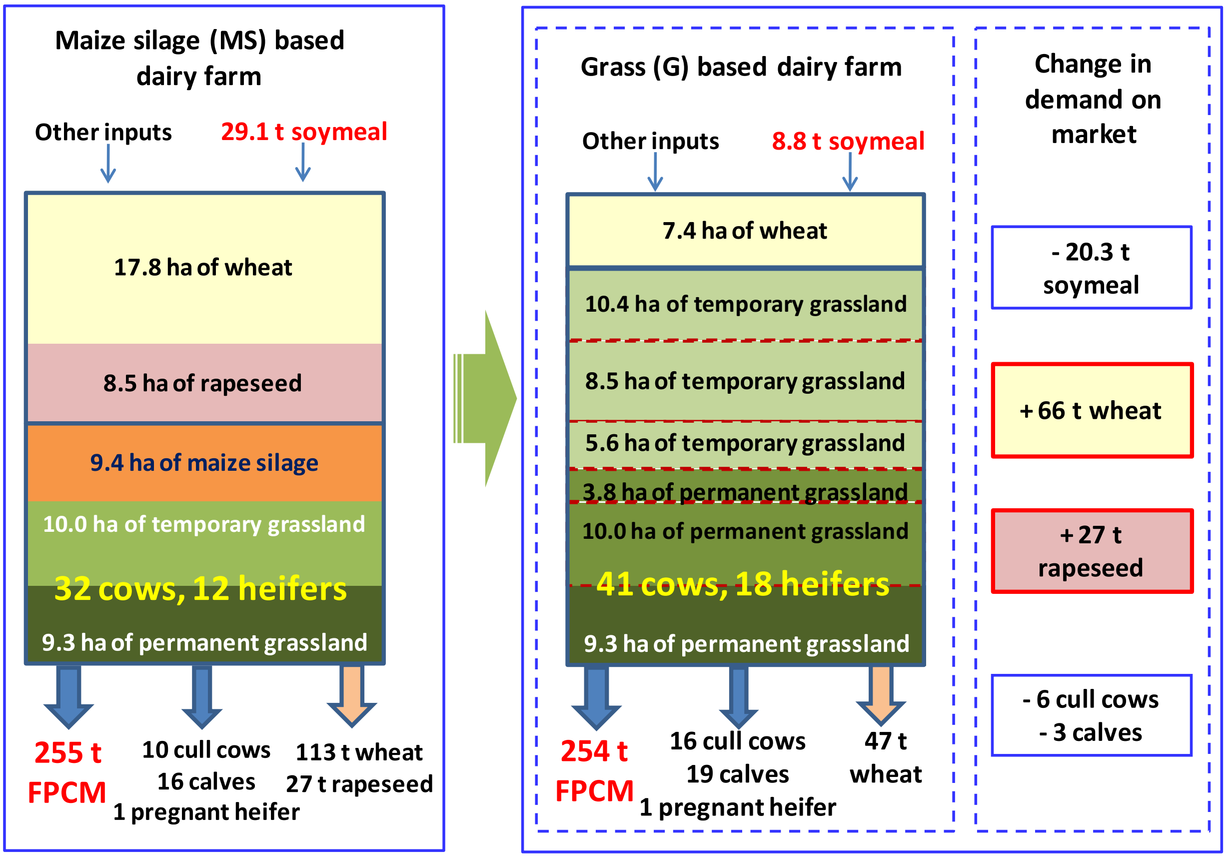

Figure 1: Consequences of converting a maize-silage-based dairy farm to a grass-based dairy farm while keeping on-farm area and milk production constant

In each dairy subsystem, we assumed that 8% of the milk produced is not sold because it is fed to calves or lost to diseases such as mastitis. Calves not kept for replacement are sold after 2 weeks (55 kg live weight, LW). We assumed that one heifer died at the end of the first year. Heifers are raised up to their first calving, but those not incorporated into the cow herd are sold as pregnant heifers. We assumed that one cull-cow died; the rest are fattened with grass silage and concentrate for two months before slaughter. We assumed that losses during grazing or conservation and feeding of grass are about 15% of dry matter (DM) produced, and losses during conservation and feeding of maize silage are about 10% of DM produced. Cows are raised indoors in a loose housing system, and slurry from the feeding area is evacuated and stored outside the animal housing. The bedding area for lactating cows is covered with straw, and solid manure is collected indoors and removed every 2 months. Other types of animals are housed in deep bedding, and manure is collected indoors and removed once per year. Table 1 describes the two dairy farms; further details are presented in Chapter 4.

Table 1: Characteristics of the maize-silage-based dairy farm (before farm conversion) and grass-based dairy farm (after conversion)

	Unit	Maize-silage-based	Grass-based
On-farm area	Ha	55.0	55.0
Temporary grassland	Ha	9.3	23.1
Permanent grassland	Ha	10.0	24.5
Silage maize	Ha	9.4	0.0
Wheat	Ha	17.8	7.4
Rapeseed	Ha	8.5	0.0
Herd characteristics			
Cows	Animals	32	41
Heifers (0-1 yr)	Animals	13	19
Heifers (1-2 yr)	Animals	12	18
Heifers (2-3 yr)	Animals	0	18
Replacement rate	%	34	41
Cull rate	%	31	39
FPCM[1]	t/cow/yr	8.66	6.74
Concentrate feed fed			
Wheat	t dry matter	11.4	13.2
Soybean meal	t dry matter	29.1	8.8
Minerals	t dry matter	3.9	4.4
Dairy farm outputs (sold)			
FPCM[1]	T	254.9	254.2
Finished cull cows	t live weight	7.4	11.8
Weaned calves	t live weight	0.9	1.1
Pregnant heifers	Animals	1	1
Wheat	t dry matter	113.5	47.2
Rapeseed	t dry matter	27.4	0.0

[1]Fat and protein corrected milk

2.2. Consequences of farm conversion

The consequences of an action can propagate through global economic and technological systems in chains of cause-and-effect relationships (Ekvall and Weidema, 2004). Thus, CLCA represents a convergence of LCA and economic modelling methods (Earles and Halog, 2011).

In this study, we investigated the consequences of increased preference for grass-based milk. We based this work on methodologies proposed by Schmidt (2008), Edwards et al. (2010) and Smyth and Murphy (2011) to identify affected processes (i.e. those expected to be affected most by the increased preference) and possible consequences due to cause-and-effect relationships.

2.2.1. Change in herd structure and meat production

When the conversion from MS to G occurs, 9 cows and 6 heifers must be added to the herd to maintain the same milk production. We assumed these animals were purchased at conversion. For simplicity's sake, we did not consider impacts associated with these animals, as this purchase only occurs once (at the transition) and this additional stock will be available once the farm is sold or ended. We thus assumed a zero-sum situation. The G herd also produces 3 more calves and 6 more cull cows per year than the MS herd, which we assumed to decrease demand, and therefore production, of suckler beef cattle. Avoided impacts of suckler beef cattle production were taken from Nguyen et al. (2012) (chapter 2).

2.2.2. Change in on-farm cropping pattern and rotation and their influence on C stock of farm soil

To satisfy forage requirements for the G herd, on-farm cropping pattern and its associated rotation change (Table 2). We assumed that all temporary grassland and some of the silage maize area on the MS farm are converted to permanent grassland on the G farm, while the rest of the silage maize area, the entire rapeseed area and some of the wheat area are converted to temporary grassland. Thirty years of C dynamics in the topsoil (0-30 cm) was simulated for each rotation of each farm with the model RothC 26.3 (Coleman and Jenkinson, 1996). RothC has been used and tested extensively under various climates and agricultural contexts, including grasslands (Smith et al., 1997). Input variables of RothC include initial C stock; monthly rainfall, temperature and evapotranspiration; amount of plant cover; soil clay content; and organic C inputs (e.g., plant residues, organic fertilisers). These variables were estimated for MS and G farms in Normandy, France. Table 2 shows predicted changes in soil C stocks due to conversion-induced changes in cropping patterns.

Table 2: Change in cropping pattern and rotation due to the conversion from maize-silage-based dairy farm (MS farm) to grass-based dairy farm (G farm) and consequences on average annual C stock increase of on-farm area simulated with RothC over a 30-year period.

Cropping pattern of MS farm	Rotation of MS farm		Cropping pattern of G farm	Rotation of G farm	Increase in C stock due to farm conversion (t C/ha/yr)	Area (ha)
Permanent grassland	PG	→	Permanent grassland	PG	0.29	9.3
Temporary grassland	TG5/M1/W1	→	Permanent grassland	PG	0.46	10.0
Silage maize	TG5/M1/W1	→	Permanent grassland	PG	0.46	2.0
Silage maize	M1/W1	→	Permanent grassland	PG	0.88	1.8
Silage maize	M1/W1	→	Temporary grassland	TG5/W2	0.48	5.6
Wheat	TG5/M1/W1	→	Temporary grassland	TG5/W2	0.06	2.0
Wheat	M1/W1	→	Wheat	TG5/W2	0.48	7.4
Wheat	R1/W1	→	Temporary grassland	TG5/W2	0.35	8.4
Rapeseed	R1/W1	→	Temporary grassland	TG5/W2	0.35	3.8
Rapeseed	R1/W1	→	Temporary grassland	TG5/TG5	0.35	1.2
Rapeseed	R1/W1	→	Temporary grassland	TG1/TG1	0.11	3.5

PG: Permanent grassland
TG: Temporary grassland
M: Silage maize
W: Wheat
R: Rapeseed
The number after the type of crop or grass indicates its duration (in years) in the rotation.

2.2.3. Consequences of decreased on-farm cash-crop production

As most of the cash-crop area is converted to grassland, less wheat and rapeseed are provided to the international market, which increases their prices; this lowers demand for them and encourages increased production (Kløverpris et al., 2010). Production of a crop can increase in three ways: increasing yields in existing fields via intensification, replacing other crops with the desired crop (crop displacement), and expanding production onto other land use types (e.g., forest, grassland), which causes land use change (LUC) (Edwards et al., 2010; Kløverpris et al., 2010). In reality, many causal chains of crop displacement will occur simultaneously in response to a changing supply of or demand for crops; to assess them, a simulation model of world agricultural markets is essential (Edwards et al., 2010). We therefore used results of Edwards et al. (2010), who predicted effects of increased biofuel demand on LUC with several economic models. Using such models requires the strong assumption that changes in international markets respond linearly to the magnitude of crop demand.

To analyse the sensitivity of LUC predictions to models and their parameters, we used results from the Global Trade Analysis Project (GTAP, version GTAP-BIO) and the Landbouw Economisch Instituut Trade Analysis Project (LEITAP, version LEITAP2) models (Edwards et al., 2010). GTAP is a multi-region, multi-sector general equilibrium model based on neoclassical economic theory, in which prices adjust to create an equilibrium between supply and demand of all goods, services, and production factors in the economy (Kløverpris et al., 2010). The version GTAP-BIO was designed specifically to analyse global impacts of expanded biofuel production (Edwards et al., 2010). The modified database used in GTAP-BIO was based on version 6 of the GTAP database, which represents the global economy with 87 regions and 57 sectors each, addresses biofuel by-products better, and estimates global land use more accurately (Edwards et al., 2010). The LEITAP model, based on GTAP, was developed to analyse impacts of the EU biofuel directive on agricultural markets. The version LEITAP2 adds a land-supply curve based on biophysical model outcomes from IMAGE (Eickhout et al., 2007) and Dyna-CLUE (Verburg et al., 2008) to predict LUC and distinguishes between marginal and average land productivity (Edwards et al., 2010).

Both models account for LUC area due to changes in by-products of biofuel production, food consumption, and crop yields. Edwards et al. (2010) reported results as LUC area per tonne of oil equivalent (toe) of biofuel and the contribution of each of the three factors. In our study, the decrease in supply of crop products of G farm relative to MS farm results from the conversion of cropland to grassland, hence no crop is produced on this land, and no by-products are produced. Therefore, we did not consider the effect of by-products on the reduction in LUC.

From Edwards et al. (2010), we used results from the GTAP scenarios “Marginal extra ethanol from wheat demand in EU” (EU wheat ethanol) and “Marginal extra biodiesel from oilseed demand in EU” and the LEITAP scenarios “Increase in demand of ethanol from wheat in France” and “Increase in demand of biodiesel in Germany”. In both GTAP and LEITAP scenarios, LUC due to increased biodiesel demand results from the production of the oilseed mix used to produce biodiesel. For our study, we assumed that LUC in biodiesel scenarios was caused only by rapeseed production. Table 3 presents LUC predictions from GTAP and

LEITAP (Edwards et al., 2010) and the results used in our study (ignoring the use of by-products).

For example, the GTAP scenario “EU wheat ethanol” predicted that production of 1 additional toe ethanol requires 5.2 additional t of wheat as feedstock (Table 3). However, wheat requirements decrease to 3.6 t due to the use of by-products from ethanol production (32%) and then decrease further to 1.9 t due to a reduction in human consumption (46%). Assuming a yield of 5.5 t/ha, producing this amount of wheat requires an area of 0.34 ha, from which an estimated 0.03 ha can be subtracted by increasing wheat yields. Then the remaining 0.31 ha area can be partially “saved” by crop displacement. In this case, crop displacement resulted in an increase in demand for additional land by 0.48 ha. Thus, the net additional area needed is (0.31 + 0.48 =) 0.79 ha worldwide, which will come from forest and grassland. In fact, producing 0.79 additional ha of wheat requires producing 1.11 ha of wheat, 0.01 ha of oilseeds, 0.04 ha of other crops and stopping production of rice and coarse grain (i.e. cereal grains other than wheat and rice, used primarily for animal feed or brewing) on 0.36 ha (Table 4). Similarly, according to GTAP, without considering by-products, marginal demand for 1 t of wheat requires a net amount of 0.54 t of wheat to be produced, resulting in 0.22 ha of LUC. In the same way, marginal demand for 1 t of rapeseed requires the production of a net amount of 0.99 t of rapeseed, resulting in 0.33 ha of LUC. For LEITAP, without considering by-products, marginal demand for 1 t of wheat requires a net production of 0.97 t of wheat, resulting in 0.13 ha of LUC. Similarly, marginal demand for 1 t of rapeseed requires a net amount of 0.91 t of rapeseed, resulting in 0.65 ha of LUC.

Table 3: Land use change (LUC) key model parameters for GTAP and LEITAP models (Edwards et al., 2010) and their application in the present study for increases in demand of wheat and rapeseed due to farm conversion.

Model/ Scenario used and application in this study	Gross demand of feedstock (t)		Reduction factor by using by-products (%)		Food consumption reduction factor (%)		Net demand of feedstock (t)		Average yield (t/ha)		Area without yield savings (ha)		Area "saved" by yield increase (ha)		Area "saved" by crop displacement (ha)		LUC (ha)
GTAP																	
EU wheat ethanol[1]	5.2	-	32%	-	46%	=	1.92	÷	5.5	=	0.34	-	0.03	-	- 0.48	=	0.79
EU biodiesel (mix)[2]	2.4	-	52%	-	1%	=	1.14	÷	5.5	=	0.21	-	0.25	-	- 0.42	=	0.38
GTAP applied in this study																	
Demand for wheat	1.0	-	0%	-	46%	=	0.54	÷	5.5	=	0.10	-	0.01	-	- 0.14	=	0.22
Demand for rapeseed	1.0	-	0%	-	1%	=	0.99	÷	5.5	=	0.18	-	0.22	-	- 0.36	=	0.33
LEITAP																	
Wheat ethanol Fra[3]	5.5	-	1%	-	3%	=	5.28	÷	4.2	=	1.26	-	0.15	-	0.38	=	0.73
Biodiesel Deu[4]	3.0	-	1%	-	9%	=	2.70	÷	4.2	=	0.64	-	0.36	-	- 1.64	=	1.93
LEITAP applied in this study																	
Demand for wheat	1.0	-	0%	-	3%	=	0.97	÷	4.2	=	0.23	-	0.03	-	0.07	=	0.13
Demand for rapeseed	1.0	-	0%	-	9%	=	0.91	÷	4.2	=	0.22	-	0.12	-	- 0.55	=	0.65

[1]Marginal extra ethanol from wheat demand in EU
[2]Marginal extra biodiesel from oilseed demand in EU
[3]Increase in demand of ethanol from wheat in France
[4]Increase in demand of biodiesel in Germany

Table 4: Worldwide climate change impact of increased marginal crop production demand according to the two economic models applied.

	Paddy rice	Wheat	Coarse grains[1]	Oilseeds	Sugar crops[2]	Other crops	Marginal crop[3]
Increase in demand for wheat/GTAP							
Area (ha)	-0.03	1.11	-0.33	0.01	0	0.04	0.79
Climate change (t CO_2/ha)[4]	10.14	2.17	2.75	0.50[5]	1.82	0.00	1.58
Increase in demand for rapeseed/GTAP							
Area (ha)	-0.02	-0.05	-0.02	0.47	0	-0.05	0.33
Climate change (t CO_2/ha)[4]	10.14	2.17	2.75	0.95[6]	1.82	0	0.20
Increase in demand for wheat/LEITAP							
Area (ha)	0	1.13	-0.32	-0.16	0.07	0.01	0.73
Climate change (t CO_2/ha)[4]	10.14	2.17	2.75	0.96[6]	1.82	0.00	2.19
Increase in demand for rapeseed/LEITAP							
Area (ha)	0	-0.03	-0.35	2.34	-0.02	0	1.93
Climate change (t CO_2/ha)[4]	10.14	2.17	2.75	0.91[7]	1.82	0.00	0.55

[1]Coarse grain: assumed to consist of a 5:1 ratio of maize and barley, based on 2003-2007 worldwide production (FAOSTAT, 2012).
[2]Sugar crops: assumed to consist of a 6:1 ratio of sugar cane and sugar beet, based on 2003-2007 worldwide production (FAOSTAT, 2012).
[3]Marginal crop: assumed as the sum of all precedent crops
[4]From Nemecek et al. (2011)
[5]Oilseeds: assumed to consist mainly of soybean, according to model predictions of regional distribution.
[6]Oilseeds: assumed to consist mainly of a 1:2 ratio of soybean and rapeseed, according to model predictions of regional distribution.
[7]Oilseeds: assumed to consist mainly of a 1:1.5 ratio of soybean and rapeseed, according to model predictions of regional distribution.

Average worldwide climate change impact data (not including LUC) associated with the production of principal crops (wheat, rice, maize, barley, sugar crops, rapeseed and soybean) was taken from Nemecek et al. (2011) (Table 4). We assumed that coarse grain consisted of a 5:1 ratio of maize and barley and that sugar crops consisted of a 6:1 ratio of sugarcane and sugar beet based on worldwide production from 2003-2007 (FAOSTAT, 2012). We assumed that oilseeds consist mainly of soybean (EU wheat ethanol/GTAP) or of a mix of soybean and rapeseed (for other scenarios) according to the regional distribution predicted by the models. Climate change impacts associated with production of the marginal crop took crop displacement into account (Table 4). Only the GTAP model provided GHG emission factors for regional LUC (Table S1), but we applied them for both GTAP and LEITAP. Table 5 summarises worldwide conversion between land use types (e.g., grassland to forest, forest to cropland, and grassland to cropland) due to the increase in crop demand and C release/sequestration associated with it predicted by the models and applied to our case studies.

Table 5: Land use change (LUC) emissions (over a 30-year period) due to a gain of 1 ha of cropland and resulting conversion of different land use types, according to the two economic models applied.

	Loss of forest (ha)	Emissions from forest loss (t CO_2/yr)	Forest gain from grassland (ha)	Sequestration from forest gain (t CO_2/yr)	Loss of grassland (ha)	Emissions from loss of grassland (t CO_2/yr)	Gain in cropland (ha)	Sequestration from gain in cropland (t CO_2/yr)	Emissions per ha of LUC (t CO_2/ha/yr)
Applied GTAP in this study									
Increase in demand for wheat	-0.44	-5.7	0.30	2.8	-0.86	-3.2	1.00	0.6	-5.4
Increase in demand for rapeseed	-0.41	-5.0	0.28	2.7	-0.87	-3.1	1.00	0.6	-4.8
Applied LEITAP in this study									
Increase in demand for wheat	-0.89	-12.2	0.03	0.2	-0.14	-0.7	1.00	0.6	-12.1
Increase in demand for rapeseed	-0.88	-12.7	0.00	0.0	-0.12	-0.5	1.00	0.6	-12.6

2.2.4. Choice of data used

We assumed that conversion from an MS to a G farm does not change consumption of inputs such as electricity, fuel, fertilisers, minerals and purchased wheat, and will not affect market demand; thus, we used average data for them. In contrast, conversion decreases soybean meal consumption by 20.3 t (Table 1). While the MS farm used an average soybean imported from Brazil, a major part of which is associated with deforestation (Prudêncio da Silva et al., 2010), we assumed that soybean meal bought by the G farm was imported from a region without deforestation (i.e., southern Brazil). We thus suppose that the reduced demand allows procuring soybean meal not associated with deforestation.

2.3. Life cycle impact assessment

The environmental impacts of the entire MS farm and its milk, meat and cash-crop production were as presented in Chapter 4. The impacts of surplus animals for the G farm (relative to MS farm) and of sold animals (for both farms) were determined with system expansion, in which impacts of dairy cattle were assumed to equal those of suckler-beef cattle estimated by Nguyen et al. (2012). The environmental consequences of farm conversion were attributed to the milk, as it drove the decision to convert. The impacts of wheat and rapeseed produced outside the G farm have two components: 1) average world-wide climate change impacts (expressed per ha) according to Nemecek (2011) and 2) LUC and associated GHG emissions as calculated using GTAP and LEITAP; total impacts are higher than those of wheat and rapeseed produced on the MS farm. Differences between impacts of wheat and rapeseed produced outside the G farm vs. those of wheat and rapeseed produced on the MS farm were attributed to G milk. Impacts were expressed for the whole farm and per 1 t of FPCM. The impact categories considered were climate change including the effects of land use and land use change (CC/LULUC) (kg CO_2 equivalent (eq.)) and land occupation (LO) (m^2*yr). The impacts CC/LULUC and LO were calculated using the CML2 “baseline” and “all categories” 2001 characterisation methods as implemented in the ecoinvent v2.0 database.

3. Results

The environmental impacts of the entire G farm, including effects of farm conversion can be expressed as the sum of four components: a) impacts of production on the G farm itself (including avoided impacts due to avoided production of soybean meal), b) C sequestration due to on-farm LUC, c) avoided impacts due to production of more surplus animals than on the MS farm, and d) indirect impacts due to additional wheat and rapeseed produced outside the G farm (Table 6). On-farm LUC (conversion of cropland to grassland and of temporary to permanent grassland) sequesters C, which reduces CC/LULUC of the G farm itself by 15%. Additional surplus animals from the G farm reduce impacts of G farm (avoided impacts) by 26 and 15% for LO and CC/LULUC, respectively. The production of additional wheat and rapeseed outside the G farm induces an increase in its impacts of 40 and 45% for LO, 29 and 69% for CC/LULUC with GTAP and LEITAP, respectively. The impacts with GTAP are lower than with LEITAP for CC/LULUC of wheat (by -18%) and rapeseed (-81%), and for LO of rapeseed (-49%), except for LO of wheat (higher by 65%). Compared to LEITAP, the use of GTAP for estimating LUC associated with additional wheat and rapeseed yields values for LO and CC/LULUC of the G farm that are 4 and 29% lower, respectively.

The conversion increases LO and CC/LULUC of the G farm, respectively, by 9 and 7% with GTAP and by 14 and 51% with LEITAP (Table 7). As impacts of all conversion consequences are attributed to milk, LO and CC/LULUC of G milk increase, respectively, by 82 and 13% with GTAP and by 123 and 98% with LEITAP. Of whole-farm LO and CC/LULUC impacts, milk production contributes, respectively, 19 and 56% with GTAP and 23 and 69% with LEITAP for the G farm, compared to 12 and 53% for the MS farm (Table 7). Therefore, LO and CC/LULUC per 1 t of grass-based FPCM are, respectively, 0.05 ha*yr and 1.12 t CO_2 eq. with GTAP and 0.06 ha*yr and 1.95 t CO_2 eq. with LEITAP, whereas per 1 t of maize-silage-based FPCM, they are 0.03 ha*yr and 0.96 t CO_2 eq., respectively. The impacts of wheat and rapeseed produced outside the G farm due to the consequential increase in crop production differ from those produced on the MS farm by respectively +39 and +6% for LO and +239 and +52% for CC/LULUC with GTAP and by respectively -16 and +108% for LO and +314 and +687% for CC/LULUC with LEITAP (Table 8).

Table 6: Effects of the conversion from a maize-silage-based dairy farm (MS farm) to a grass-based dairy farm (G farm) on land occupation and climate change impact of the latter, according to the two economic models applied.

	Effects of farm conversion							Total impacts of G farm (after conversion)[1]	
	G farm production itself	C sequestration due to on-farm LUC	Avoided surplus animals	Additional wheat /GTAP	Additional rapeseed /GTAP	Additional wheat /LEITAP	Additional rapeseed /LEITAP	GTAP	LEITAP
Land occupation (ha*yr)	59.7		-15.7	14.8	9.0	8.9	17.8	67.9	70.8
Climate change (t CO_2 eq)	516.3	-77.9	-77.3	103.5	45.0	126.4	233.6	509.6	721.0

[1]The G farm produces the same amounts of outputs as the MS farm

Table 7: Impacts for the whole farm and for milk production of a maize-silage-based dairy farm (MS farm) and a grass-based dairy farm (G farm) after conversion, and increase in impacts due to conversion expressed as a percentage of those of the MS farm, according to the two economic models applied.

	Before farm conversion (MS farm)				After farm conversion (G farm[1])				Increase (%) of impacts due to conversion			
	Whole MS farm	Milk prod. MS farm	Meat prod. MS farm	Cash-crop prod. MS farm	Whole G farm /GTAP	Whole G farm /LEITAP	Milk prod. G farm /GTAP	Milk prod. G farm /LEITAP	Whole farm /GTAP	Whole farm /LEITAP	Milk prod. /GTAP	Milk prod. /LEITAP
Land occupation (ha*yr)	62.0	7.1	29.2	25.7	67.9	70.8	13.0	15.9	9	14	82	123
Climate change (t CO_2 eq)	476.1	250.8	146.2	79.0	509.6	721.0	284.3	495.8	7	51	13	98

[1]The G farm produces the same amounts of outputs as the MS farm

Table 8: Environmental impacts of wheat and rapeseed produced on maize-silage-based dairy farm (MS farm) or produced outside of grass-based dairy farm (G farm) due to the farm conversion according to GTAP and LEITAP models

	Wheat on MS farm[1]	Rapeseed on MS farm[1]	Wheat outside G farm /GTAP	Rapeseed outside G farm /GTAP	Wheat outside G farm /LEITAP	Rapeseed outside G farm /LEITAP
Land occupation (ha*yr/t)	0.16	0.32	0.22	0.33	0.14	0.66
Climate change (t CO2 eq./t)	0.46	1.10	1.57	1.67	1.91	8.65

[1]Results from Chapter 4

4. Discussion

4.1. Methodology of CLCA: from principle to implementation

Application of CLCA modelling requires introducing market mechanisms via affected processes (Zamagni et al., 2012), but identifying the affected processes remains a challenge (Dalgaard et al., 2008, Zamagni et al., 2012). Schmidt (2008) proposed a procedure to identify affected processes in agricultural CLCA, especially of crop production, using a step-wise approach introduced by Weidema et al. (1999). Schmidt (2008) proposed that increased demand for a certain crop can be met by an increase in yield and/or in production area by crop displacement or transformation of non-agricultural land into agricultural land. However, a limitation of this procedure is that crop substitutions were assumed to occur within product types (e.g. one cereal is used instead of another cereal, one oilseed instead of another oilseed) (Zamagni et al., 2012). Currently, products and markets are highly connected on a global scale; thus, increased demand for a product may not only increase its price but also decrease consumption of other products produced elsewhere in the world. Therefore, the use of economic models developed to predict global economic mechanisms is useful and recommended to identify affected processes in CLCA.

Economic models recently have been used to predict changes in land use (Kløverpris et al., 2008; 2010) and GHG emissions due to increased demand for biofuels (Verburg et al., 2009, Edwards et al., 2010). For livestock production systems, the only known study applying CLCA is by Thomassen et al. (2008), who investigated environmental consequences of increased milk demand. This increase in demand induced an increase in milk production and hence, authors assumed that at least one more dairy farm was needed. Thomassen et al. (2008) identified affected processes as electricity (production from natural gas), feed protein (soybean meal from Argentina) and feed energy (spring barley). However, they did not investigate how to meet the increased demand for area (in particular grassland area) required for milk production by an additional farm and its LULUC consequences.

Smyth and Murphy (2011) investigated the indirect effects of increased biomethane production from grass on the livestock sector in Ireland using a causal-descriptive method, whereby cause-and-effect logic was used to predict system behaviour and define indirect consequences (i.e. affected processes). They assumed that grass biomethane would be produced at the expense of Irish beef production, thus decreasing beef exports to the UK, leading the UK to import more meat from other countries. They did not, however, investigate the consequences of increased meat production in other countries. Although the chain of consequences resulting from a change in demand may seem a never-ending story (Weidema, 1999), CLCA practitioners should try to consider all consequences up to and including LUC.

In our study, we examined one possible way to meet increased preference for G milk: converting the base of feeding systems on domestic dairy farms from silage maize to grass. Other ways to meet this increase in demand (e.g. importing or increasing yields of G milk; Fig. 2) need to be investigated. We assumed that French consumers would purchase the same quantity of G milk as they did MS milk, regardless of price. However, increasing demand may further increase its price, which may decrease its consumption. As a result, total milk consumption might decrease for a certain time, during which the amount of additional G milk produced would be smaller than the amount of MS milk it replaced.

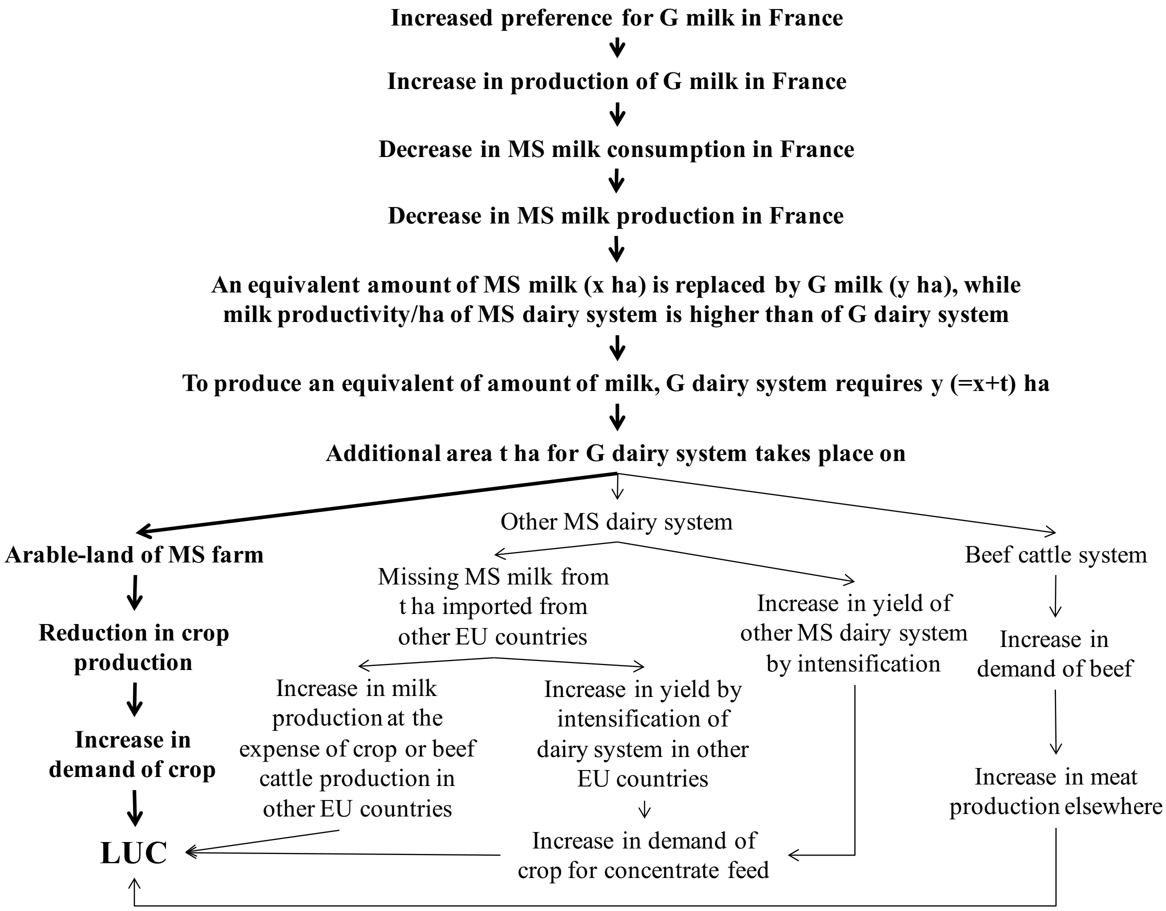

Figure 2: Possible consequences of increased preference for grass-based (G) milk in France. Cause and effect chain explored in the paper is in bold.

MS milk: maize-silage-based milk
G dairy system: grass-based dairy system
MS dairy system: maize-silage-based dairy system
MS farm: maize-silage-based dairy farm, including MS dairy system and cash crop system
LUC: land-use change

Another simplification in our study was the assumption that the additional beef produced by the G farm would replace meat produced by a suckler-beef system. In addition, the increase in beef production due to dairy-farm conversion could reduce beef price and, consequently, consumption of other types of meat as well (not only beef from suckler cattle). Kløverpris et al. (2010) used GTAP to predict effects of increased household wheat consumption (resulting in a higher price) on LULUC. They estimated that due to limited household budgets, increased wheat consumption would be balanced by an equal decrease (in monetary terms) in consumption of other commodities.

Our study, contrary to many others, included the consequences on LULUC impacts. First, the G farm replaces a MS farm, which affects the on-farm cropping pattern and crop management practices and hence on-farm soil C dynamics. Secondly, the G farm requires more land to produce the same quantity of milk as the MS farm. Thus, the area used for cash-crop production decreases, which subsequently shifts demand in the market to other sources. These effects were accounted for by using GTAP and LEITAP simulation results as reported by Edwards et al. (2010). Application of the GTAP and LEITAP models seemed justified by the fact that these models are either linear or nearly so, i.e. increases in crop area are roughly proportional to the extra demand for a particular crop (Edwards et al., 2010). Because these models cannot simulate more than one crop at a time, we assumed that the farm conversion consequences on LUC due to the increase in demand of wheat and rapeseed are additive.

4.2. Co-product handling

Once consequences of farm conversion are defined, the difficulty remains of how to attribute the effects of these consequences to several products. Although increased preference for G milk drives farm conversion, one wonders whether G milk should be held responsible for the impacts of subsequent consequences. As dairy production systems produce both milk and beef, some impacts associated with these consequences could be attributed either to all beef or just the additional beef (compared to that from the MS farm) produced by the G farm. By estimating impacts of G beef as equal to those of suckler beef (using system expansion), some impacts of farm conversion were attributed exclusively to beef. This is because the suckler-beef-cattle system is also based mainly on grassland (like beef meat from G farm), and its beef has been shown to have higher environmental impacts than dairy beef (when allocation methods were applied for milk and its co-product dairy beef) (Chapter 4).

4.3. Effects on LULUC

The use of economic models is appropriate for taking into account indirect effects of farm conversion on LUC (Prins et al., 2010). However, the predictions of these models have high variability and uncertainty due to their characteristics, hypotheses and assumptions (Laborde 2011, Marelli et al., 2011). GTAP and LEITAP predictions for wheat and rapeseed clearly reveal this variability, which resulted mainly from the way that change in surface area is calculated per crop per region. LEITAP assumes that the area of crop expansion depends strongly on the average yield of the particular crop whose production increases, whereas GTAP assumes that it depends on the yield of that same crop at the frontier of cultivation, i.e. in the region where LUC occurs. Thus, GTAP includes a factor which estimates the yield at the frontier of crop production, where LUC resulting in marginal increases in agricultural area actually occurs (Edwards et al., 2010).

The models also differ in the way that additional production is shifted from countries with high yields to less-developed countries with lower yields (Edwards et al., 2010). Other parameters differ, such as the factor increasing yield with price, and the reduction factor due to by-product use or food consumption (Edwards et al., 2010, Prins et al., 2010). In our study, the same gross demand for feedstock yielded large differences in predicted LUC and its emissions in the two models, despite not using the by-product reduction factor and using the GTAP emission factor for LUC for both models. Compared to GTAP, LEITAP predicted increases in LUC area and emissions up to 2 and 5 times larger, respectively. In LEITAP, nearly 90% of LUC is converted from forest and the rest from grassland into cropland, whereas in GTAP nearly 90% LUC is converted from grassland into cropland. In addition, in GTAP nearly 68% of the total forest loss is converted to grassland (only up to 3% in LEITAP).

4.4. Uncertainty

In CLCA modelling, the high degree of uncertainty and wide range of possible results depend on choices with respect to: 1) system enlargement (and thus on the affected processes taken into account), 2) the indirect effects included and 3) the hypotheses, assumptions and scenarios considered (Zamagni et al., 2012). Uncertainty is also a critical issue in estimating the effects of LUC (Edwards et al., 2010, Prins et al., 2010, Laborde 2011, Marelli et al., 2011). Apart from model characteristics and assumptions used to predict LUC, uncertainty also results from a crop’s estimated increase in yield with price and future trade (Prins et al., 2010) and assumptions about how its production will shift among regions and on what type of land (forest, grassland, cropland) it will be produced. Last but not least, estimation of CO_2 emissions due to crop expansion on forest or grassland is also highly uncertain.

In our study, assessment of the impacts of the on-farm production system is probably less uncertain than the estimated consequences outside the farm due its conversion. First, as described in Chapter 4, enteric CH_4 was estimated with the Tier-3 method used in the French Inventory of Greenhouse Gases (Vermorel et al., 2008), as recommended by IPCC (IPCC, 2006) to improve the accuracy of emission estimates. Our data on DM intake, milk production, herd management and farming practices were taken from a systems experiment (for G farm, Delaby et al., 2009) and a farm network followed by the French Livestock Institute (for MS farm, Pavie et al., 2010), which can be assumed to have low uncertainty. Our estimates of N_2O emissions are based on IPCC Tier-2 (IPCC, 2006) emission factors, which have relatively high uncertainty.

Regarding on-farm soil C dynamics, for sensibility analysis we also simulated a 100-year period after farm conversion with RothC, in addition to the 30-year period. During the 100-year period, predicted on-farm soil C sequestration was 45.7 t CO_2/year, 43% lower than the 79.7 t CO_2/year during the 30-year period, indicating that C will continue to be sequestered after the first 30 years after conversion, but at a lower rate. If considering a 100-year period, total climate change impact of the G farm (after conversion) would be 7% and 5% higher than its impact estimated with a 30-year period for GTAP and LEITAP, respectively (results not shown). So, the timeframe for estimating soil C dynamics is an important factor contributing to uncertainty.

Ultimately, we investigated only one of the many potential cause-and-effect pathways of farm conversion, which limits the study's ability to quantify uncertainty, despite using two models to estimate LUC consequences. Scenario analysis of other potential consequences of this type of farm conversion would enrich further studies.

4.5. Consequences of increased preference for grass-based milk

This study has illustrated how an increase in preference for G milk could be met and explored its potential environmental consequences. Without considering the consequences of reduced on-farm production of cash-crops which occur outside the G farm boundary (i.e. indirect consequences), climate change and land use impacts of the G farm were substantially lower than those of the MS farm (by 24 and 29%, respectively). Including indirect consequences necessary to compensate for decreased outputs from the G farm greatly increased impacts of the G farm, in function of how indirect effects were estimated. Therefore, for such a conversion, it would be misleading to only look at what happens at the on-farm level, because of the major indirect effects outside the farm.

In our study, C sequestration resulting from on-farm LUC and CO_2 emissions due to LUC for additional crops were amortized over 30 years. As C sinks/sources resulting from sequestration/emission activities in soil or biomass are not permanent (e.g. Smith, 2005), we assumed that soil C reaches a new equilibrium after 30 years and remains stable. So the results presented in this paper concern the 30-year period after conversion. After this period, climate change impact of the G farm (including indirect consequences resulting from cash-crop production outside the G farm) would be slightly lower (less than 2% with GTAP and 3% with LEITAP) than that of the MS farm (results not shown). In contrast, direct and indirect land occupation of the G farm remain higher, and therefore the consequences in terms of biodiversity loss due to forest and pasture conversion into cropland remain.

5. Conclusion

Contrary to an ALCA approach, the CLCA approach allows assessing the consequences of a change in the life cycle on the processes beyond the system boundary, in particular on LUC. The integration of global economic models in CLCA modelling is necessary to identify and assess the processes affected by a change in the life cycle. This study demonstrated how environmental consequences of increased preference for grass-based milk can be assessed using the CLCA approach. Although this study investigated only one way to meet this increased preference, it indicates that the conversion of a maize-silage-based to a grass-based dairy system would have substantial consequences on LUC outside the farm and that it is important to consider these. However, scenario analysis needs to be performed in further studies to explore other possible consequences. Finally, the approach applied in this study needs to be further developed to identify the advantages and weaknesses of grassland use in ruminant feeding systems worldwide.

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Supplementary materials

Table S1: GTAP emission factors for land use change (t CO_2/ha)

Geographical entity	Emissions due to forest loss	Sequestration due to forest gain	Sequestration due to gain in crop area	Emission due to loss of pasture
European Union 27	297	362	18	156
Russia	311	392	18	156
Other CIS and CEE**	297	362	18	156
Rest of Europe	297	362	18	156
Sub Sahara Africa	305	129	18	43
Mid-East North Africa	152	59	18	82
Canada	705	434	18	196
United States of America	760	219	18	106
Brazil	388	164	18	72
Centre America Caribbean	388	164	18	72
Rest of America	388	164	18	72
Oceania	388	198	18	98
China	574	223	18	196
India	574	223	18	196
Japan	574	223	18	85
Malaysia Indonesia	937	337	18	85
East Asia	574	223	18	85
Rest of South East Asia	937	337	18	85
Rest of South Asia	937	337	18	85

* This table was adapted from Edwards et al. (2010)

** Commonwealth of Independent States and Central and Eastern Europe

Chapter 6

General Discussion and Conclusions

This general discussion focuses principally on the work presented in the thesis; however, when deemed appropriate the discussion has been enlarged to ruminant production and to livestock production in general.

6.1. Differences in ruminant production systems and impact mitigation options

Differences in environmental impacts of ruminant production systems have been analysed in Chapters 2 and 3 for suckler-beef cattle systems and in Chapter 4 for dairy systems. For the suckler-beef cattle production systems, these differences have been found to result from herd management practices for the suckler cow-calf herd and from feeding practices for the bull-fattening herd. Mainly based on grassland production, the French suckler cow-calf herd is characterised by low inputs and low cost production systems (Réseaux d'élevage Charolais, 2007; Veysset et al., 2010), which limit the options to modify diets. This also implies that it is difficult to mitigate the impacts of the suckler cow-calf herd, because implementing changes in practices is not easy for farmers and a major change in practices would be required to produce a substantial mitigation effect. As for the bull-fattening herd, the application of feeding strategies may substantially mitigate impacts, provided that animal performance is not altered. For the whole suckler-beef production system, the effects obtained for the bull-fattening herd will be diluted due to its modest contribution (5 to 20%, depending on impact categories and systems, according to our results) to the total impacts of the whole system. Obviously, these effects are considerable for farms specialised in bull-fattening (which concern only 12% of fattened animals; Office de l'Elevage, 2008).

For dairy production systems, differences may come from both herd management practices and feeding strategies. Feeding strategies are easier to apply (both for economic and practical reasons) and may produce larger mitigation effects for dairy cows than for suckler cows, due to their direct effects on milk production. Dairy production systems comprise dairy cows and other types of animals such as heifers, dry cows and cull cows for which feeding strategies are usually not suitable. Ruminant production systems often include a large grassland area, and thus grassland production and management substantially affect the overall impacts of these systems and their mitigation potential.

6.1.1. Grassland management

Both for suckler-beef and dairy production systems, grassland production was one of the most important contributors to all impacts (Chapter 2, Figure 2 and Chapter 4, Figure 2). This was because grass and grass-based forages were the main feed sources for ruminant production systems (providing 67 to 93% of total dry matter (DM) intake of the herd, both for suckler-beef and dairy systems, except for the dairy system with more than 30% of maize silage in the forage area where grass provide 46% of total DM intake). This highlights the importance of analysing opportunities to reduce the impact of ruminant production systems though grassland production.

The options for impact mitigation through grassland management have been assessed for suckler-beef production systems (Chapter 3). The potential for grassland management improvement were low because suckler-beef production systems are based on (1) a high proportion of permanent grassland (88% of the grassland area), (2) temporary grassland with ryegrass and clover mixture and (3) a low fertiliser rate for both permanent and temporary grassland (about 30 kg N/ha/yr). The mitigation options for suckler-beef production systems were (1) a small reduction of the N fertiliser rate for the grazing area and (2) a reduction of grass losses (grass not ingested by animals) during grazing. In spite of a small reduction in impacts of suckler beef production systems, a reduction of grass losses decreased the grassland area required, which can substantially contribute to impact mitigation if alternative land-use is considered.

We expect that the potential for impact reduction through grassland management will be higher for dairy production systems than for suckler-beef production systems, this would be an interesting subject for further studies. Optimization of mineral N fertiliser use without altering grass yield and a larger proportion of grassland associating ryegrass and clover can be suggested as mitigation practices for dairy systems. The use of one-year leys with ryegrass to complete crop and forage rotations in grass-based dairy farms is questioned, although this is a minor practice. Increasing the duration of this type of grassland could contribute to reducing tillage and sowing operations (i.e. reduce fuel and machinery use) and enhancing soil C sequestration.

Technical measures to reduce energy use for grassland management can be considered as an important way to mitigate the impact of both suckler-beef and dairy production system. It can be achieved by optimizing grazing area as much as possible because the production of conserved forages requires higher energy consumption than grazing (Institute de l'Elevage, 2011a). The choice of conserved forage produced also affects energy use of ruminant production systems. Among conserved forages, wrapped silage consumes the highest amount of fuel, followed by grass silage and hay (23, 15 and 12 litres/ t DM, respectively) (Institute de l'Elevage, 2011a). Moreover, the simultaneous application of several mitigation options for grassland management may achieve larger impact reductions than the use of a single option, because some options apply to several types of grassland.

6.1.2. Herd management

For both suckler-beef and dairy production systems, the largest impact reductions occurred when age of first calving was reduced from 3 to 2 years. This results from a fundamental change in herd management and subsequent feeding practices. Heifers used for replacement were reared to grow faster (by diets supplemented with concentrates) to obtain the body condition for the first breeding at 15 months and the first calving at 24 months. As a consequence, the non-productive rearing period was shortened by one year without altering productivity. For the suckler-beef production system, heifers not used for replacement also grew faster because all heifers were reared with the same management practice until their first breeding. This practice has been recently implemented in France and farmers need some time to adapt to obtain a good productivity (i.e. decreasing mortality, increasing fertility rate) for the herd (Farrié et al., 2008). This is a common practice for suckler-beef production in Canada, where suckler cows have very high productivity, breeding at 15 months, an average of 6.5 calvings per 8-year life time (Beauchemin et al., 2010). For dairy production systems, first calving at 2 years age is well adapted with Holstein breed (Le Cozler et al., 2008) and associated with an intensive management of the cows by using diets mainly based on maize silage and concentrate during the indoor period and by shortening the grazing period. As a result milk production per cow increases which contributes to reduce the impacts of the whole system. However, this intensive management also increases the risk of fertility loss and diseases (Dillon et al., 2006), which can partially counter the effects of decreased first calving age.

Changing from extensive to intensive management for heifers not used for replacement in suckler-beef systems can decrease the impacts of these animals. These heifers were kept indoors and fed a diet based on maize silage and concentrate for the entire fattening period instead of turning out during the grazing season and consuming grass-based forage in winter. These heifers grew faster due to a higher quality diet and hence finished sooner, resulting in lower impacts per unit of product (live weight gain). However, this practice moderately decreased the impacts of the whole suckler-beef production system, as these animals are only one component of a whole system. This practice requires farmers to select heifers used for replacement soon after weaning and to apply different practices with two groups of heifers instead of managing all heifers alike until the first breeding. In practice, weaned female calves which are not selected for replacement can be sold and fattened in a specialised farm. Herd management practices aimed to improve animal health and fertility and to reduce mortality are expected to reduce the impacts of production systems by reducing loss of product output. Overall, innovative herd management requires more experience and time of farmers. Such practices are not always attractive to farmers which may complicate their implementation.

6.1.3. Feeding practices

Three main feeding practices have been used: the inclusion of sources of omega-3 FA, increased supply of concentrates and the replacement of maize silage by grass silage and grazed grass. This latter practice, which is a substitution between two forages, will be analysed separately because it results in several nutritional changes, among which a decrease in cereals supplied (maize silage contains about 40% grain), and an increase in omega-3 FA.

Omega-3 FA

The influence of omega-3 FA in ruminant diets on environmental impacts was studied because they are recommended for their positive action on the nutritional quality of animal products. Increasing their concentration in ruminant diets may multiply their concentration in milk and beef by 2 and 3, respectively (Doreau et al., 2011a). It is admitted now by the scientific community that omega-3 FA have positive effects on cardiovascular diseases, infant development, and other diseases such as mental illnesses (e.g. Riediger et al., 2009). Furthermore, national nutritional recommendations are in favour of an increase in the omega-3/omega-6 ratio in human diets (PNNS, 2009 for France). Currently, dairy and meat products represent about 24 and 16% of lipid consumption in France, respectively (AFSSA, 2009), and 25 and 16% of linolenic acid consumption in France, respectively (Astorg et al., 2005).

Available data do not allow to estimate the consumption of 20- and 22-carbon omega 3 FA provided by fish and seafood (AFSSA, 2003). This shows that an increase in omega-3 FA in ruminant products can result in an improvement of the nutritional value of FA consumed by humans, as shown by Brugère-Malpuech et al. (2010). In ruminant diets, linolenic acid which is the major omega-3 FA, is present first in grass when it is grazed or conserved as silage (and to a lesser degree as wrapped silage), secondly in oleaginous seeds, mainly as linseeds which contain ca. 35% FA, more than 50% of which is linolenic acid (Doreau et al., 2012). As it has been shown that linseed FA reduce enteric methane in ruminants (Martin et al., 2010), it is especially interesting to study the effect of omega-3 FA on c limate change and other environmental impacts.

In this thesis (Chapters 2, 3 a nd 4), the effect of omega-3 FA enrichment of diets on the environmental impacts of ruminant production systems was assessed at different scales (suckler cow-calf herd and bull-fattening herd) and for two production systems (suckler-beef and dairy). For the bull-fattening herd and for the dairy cows, the increase in omega-3 FA was achieved by the use of extruded linseeds; for the suckler cow-calf herd, it was increased to a lesser extent through the replacement of hay by wrapped silage.

The reduction of the climate change impact due to omega-3 FA enrichment was clearly observed for the bull-fattening herd fed with the diet mainly based on maize silage (a 7% reduction). The effect of omega-3 FA enrichment was smaller when it was expressed at the scale of whole suckler-beef cattle production system (including the suckler cow-calf herd and the bull-fattening herd) (less than 1%) and dairy production system (1-2%). The associated effect of high concentrate proportion and omega-3 FA for the bull diet (by using a starch-based concentrate diet enriched with omega-3 FA) substantially reduced climate change for the bull-fattening herd (32%) but only resulted in a 4% reduction for the whole suckler-beef cattle system due to the small contribution (10-11%) of the bull-fattening herd to climate change for the whole system.

The modest reduction of the climate change impact for the whole suckler-beef and dairy production systems due to the use of omega-3 FA in diets can be explained by the following reasons. First, the amount of lipid added was relatively small (only 1-2%); second, the enrichment with omega-3 FA was applied for some types of animal only (e.g. fattened bulls, dairy cows) during a certain period (e.g. fattening, lactation and in-door period). Finally, the reduction in enteric methane (4.8% reduction for each percentage unit of lipid added) was partially countered by carbon dioxide and nitrous oxide emissions related to the production of omega-3 FA rich ingredients (extruded linseed, wrapped silage). However, the use of omega-3 FA in diets resulted in a small increase of other impacts of the production systems due to the production of extruded linseed (mainly due to the low yield of the linseed crop compared to cereals and to energy used for the extrusion process) and of wrapped silage (mainly due to energy use for forage preparation). Nevertheless, energy use of the suckler-beef cattle system using the starch-based concentrate diet enriched with omega-3 FA for fattened bulls strongly increased (by 10%) due to higher energy use for cereals and linseed production and concentrate feed production processes compared to maize silage production. Overall, the enrichment of ruminant diets with omega-3 FA did not substantially affect the environmental impacts studied, but it will improve nutritional quality of milk and meat products, as stated before.

Concentrate feed

Increasing concentrates in ruminant diets has often been cited as a means to reduce environmental impacts. In French dairy cow systems, a high forage intake is aimed, and the proportion of concentrate in lactating cow diets is mainly driven by milk potential and by cow voluntary intake, and depends on lactation stage. For the whole system concentrates represent 13 to 25% of DM consumed, according to the part of grass in the system (Devun and Guinot, 2012). In beef cows, that have low energy requirements compared to dairy cows, diets are always low in concentrates. High-concentrate diets are found in fattening cattle diets. In a previous trial which corresponds to a preliminary assessment of beef systems focused on the fattening period, Doreau et al. (2011b) and Nguyen et al. (2012) reported that the use of a very high concentrate diet for the bull fattening herd substantially decreased total GHG emissions and - to a smaller extent - acidification, but increased eutrophication and energy use per kg live weight gain relative to the use of diets based on hay or on maize silage. A similar result has been found in this thesis when fattening diets based on maize silage were compared to feedlot-type diets rich in concentrates based on cereals and lipids (Chapter 2). Beauchemin et al. (2011) reported that the use of a grain-based instead of a forage-based diet for pre-fattened animal before finishing in feedlot system decreased GHG emissions by 2% for the whole production system (expressed per kg carcass weight). The strong decrease in methane emission per kg DM intake with high amounts of concentrate, especially cereals, in the diet, is a well-known result (Sauvant and Giger-Reverdin, 2007; Martin et al., 2010). This is explained by a shift of the volatile fatty acid pattern towards propionate at the expense of acetate. The propionate pathway consumes hydrogen, the precursor of methane, whereas the acetate pathway produces hydrogen. In addition, increasing the proportion of cereal-based concentrate in ruminant diets improves digestibility and hence productivity (i.e. efficiency), which reduces enteric CH_4 emissions per unit of animal product. The use of concentrate feed is more economically effective for dairy cows and growing ruminants than for other types of animal in the whole system (suckler cows, replacement heifers). Again, when concentrate feed is used only for one type of animal (e.g. fattened bulls) it moderately affects the impacts of the whole production system, depending on their contribution to the impacts of the whole system. The increases in energy demand and eutrophication were due to the lower dry matter yield of feed crops, the higher energy use to produce feed crops and for the concentrate feed production process as well as due to higher nitrate leaching from feed crops than from grass production.

Increasing the proportion of cereal-based concentrate in ruminant diets to reduce enteric CH_4 and GHG emissions of ruminant production is a controversial proposition, as it ignores the ability of ruminants to convert forages (non edible for humans) into food for humans and hence increases the competition for cereals for human nutrition. This is the case if the area used for grass production is not suitable for arable crop production. In this case, forage-based ruminant production systems allow not only food production but also rural development. However, in many cases, grass is produced on land also suitable for arable crops. Further studies are needed to investigate the effects of increasing the proportion of cereal-based concentrate in diets (i.e. increasing animal productivity) and the resulting reduction in land occupation to produce the same quantity of animal product. The released land can be exploited in different ways aiming to reduce the impacts of the whole system (e.g. bioenergy, crops). In this respect, the ability of ruminants to use by-products of the bioenergy industry, several of which mainly contain fibre, needs further investigations in terms of environmental impacts. In Chapter 2, the use of a fibre-based concentrate feed (88% co-products) for fattened bulls resulted in a higher climate change impact and energy demand due to the high energy required for dehydration; impacts were even higher than with maize silage-based fattening diets. In practice, some fibre-based co-products are often dehydrated to facilitate their delivery and the incorporation in ruminant diets. Nevertheless, ruminants can convert fibre-based by-products, which are difficult to use by monogastric animals, into edible food.

Grass vs. maize silage

French dairy systems are very diverse, and differ especially in the nature of the forage used. In mountain areas, grass is the major or unique forage. In lowland areas, winter diets consist either of conserved grass or of maize silage, or of both (Devun and Guinot, 2012). Systems based on maize or on grass are often present in the same geographic area. The comparison between grass and maize silage in this thesis was based on three Holstein dairy production systems with a forage area consisting either totally of grassland or of grassland and a low (11%) or a high (33%) proportion of silage maize. These systems were first studied using the ALCA approach (Chapter 4). Changing the proportion of grass and maize silage not only results in multiple nutritional changes in the diet but affects the whole dairy system management (herd and on-farm area management). Moreover, the comparison was quite complex because it determined the level of intensification and depended on the LCA approach and co-product handling methods which will be discussed in the next sections. In general, using the ALCA approach, the impacts (except for eutrophication) per kg of fat and protein corrected milk (FPCM) of the high maize-silage system were lowest, followed by those of low maize-silage system (except with system expansion) and then those of grass-based system.

The increase in the proportion of maize silage in the annual Holstein cow diet (0, 25 and 52% maize silage in total DM intake) was associated with an increased soybean meal (0.1, 0.5 and 0.8 t/cow/year, respectively) and total DM intake (5.4, 6.4 and 7.3 t DM/cow/year, respectively) (Chapter 4). The total net energy intake increased by 27 and 40% for the diets containing maize silage, this resulted in an increase in milk yield per cow of 21 and 28%, respectively, compared to the grass-based diet. With an increasing proportion of maize silage in the diet milk yield increased, and fewer dairy cows were required to produce the same amount of milk. This partially explained the differences in impacts of milk using either grass, low or high maize silage in the diet. The reduction in the number of animals needed for the national herd due to increased milk yield per cow has substantially reduced the impacts of the dairy sector in the USA and Sweden (Capper et al., 2009; Cederberg et al., 2012).

The increased proportion of maize silage in the dairy cow diet led to more intensive herd management. This can be illustrated by the dairy system using the high proportion of silage maize. To ensure high milk production, calving was scheduled at the end of summer, the grazing period was shortened and cows were housed and fed with a diet based on maize silage. While for other systems, cows grazing continued until the end of autumn. Cows had their first calving at two years old in the system with the highest proportion of maize silage instead of 3 years in other systems. As discussed earlier, this contributed to reduce the impacts of dairy systems.

In terms of on-farm area management, dairy systems using maize silage occupied less on-farm area for the dairy herd to produce the same amount of milk because the number of animals in the herd was lower and silage maize has a higher DM yield than grassland. Thus, the available area which was not used for dairy herd was used for crop production. In addition, in the grass-based system, grassland production required a higher quantity of mineral N fertiliser to obtain a high yield of forage in order to ensure sufficient forage for the herd. This contributed to reduce the impacts of milk of the systems using maize silage relative to the system using grass only.

Nevertheless, the introduction of silage maize in the forage area increases nitrate leaching (Vertès et al., 2012). This explains why eutrophication of milk produced by low and high maize-silage dairy systems was higher than with the grass-based system. On erosion-prone soils silage maize may pose more risk to soil quality than grassland. Grassland also tends to have higher macro-scale biodiversity than silage maize (e.g., Robertson et al., 2012). Thus, milk produced from grass-based dairy system likely has lower impacts on soil quality and biodiversity than milk produced with low or high maize-silage systems.

In terms of milk quality, the increase in the proportion of maize silage resulted in a reduction in the proportion of grazed grass (51, 42 and 29%, respectively) and grass-based forages (40, 9 and 3%, respectively) in annual dairy cow diets. This reduced the omega-3 FA intake for cows because grass and grass-based forages have higher omega-3 FA content than maize silage, as mentioned in the previous section (Doreau et al., 2012). So, increasing the proportion of silage maize in grass-based dairy systems will tend to reduce the omega 3 FA content of the milk.

6.1.4. Simultaneous mitigation practices

Both the suckler-beef and dairy production systems comprise several types of animals. In addition, ruminant production systems contain other components such as grassland production, feed crop production, herd management both during grazing and indoors in winter. Generally, a mitigation practice is only applied for one type of animal or production stage, and for one component of the system. Thus the effects of such mitigation practices are diluted at the level of the whole system. In order to obtain a substantial effect for the whole system, it is necessary to apply simultaneous mitigation strategies for several types of animal and system components. Compatible practices should be combined, while considering their interaction. However, it is difficult to know the possible interactions between practices due to the lack of experimental data and appropriate simulation models.

Effects of GHG mitigation practices have been assumed to be additive (Beauchemin et al., 2011). Del Prado et al. (2010), using simulation modelling, reported that an aggregation of single-effect options tended to overestimate the effect of simultaneous mitigation practices. In practice, it is technically difficult for farmers to apply simultaneous mitigation practices, because they have to totally change their management practices, which requires time and motivation. Vellinga et al. (2011) observed that farmers prefer to apply relatively simple mitigation practices that are either cost-effective or inexpensive. However, even win-win solutions are difficult to be agreed by farmers. For example, reducing the protein content of dairy cow diets allows both a decrease in nitrous oxide, nitrate and ammonia emissions and a reduction of costs, but many farmers prefer to overfeed animals in protein as an insurance against a misevaluation of protein content of other feeds, or to increase milk production even if it is not economically sound (P. Faverdin, 2012, INRA, UMR PEGASE, Saint Gilles, personal communication). In order to motivate farmers, the attribution of subsidies not only to offset the cost of practice implementation but also to support more environmentally friendly production methods seems appropriate.

6.2. Specialisation and intensification of dairy and suckler-beef cattle sectors

The milk and beef production sectors are interlinked (Cederberg and Stadig, 2003; Zehetmeier et al., 2012). In Europe, specialised dairy production was a way to substantially increase milk production which was the priority in the 1960s to ensure food security (Pflimlin et al., 2009). Since the 1980s, milk quotas have been introduced in Europe and consequently the number of dairy cows has strongly decreased, reducing beef production from the dairy herd (Pflimlin et al., 2009). This reduction has been compensated for by an increased production of beef from suckler cows (Pflimlin et al., 2009; Institut de l'Elevage, 2011b; Cederberg et al., 2012). The reduction in beef production from specialised dairy systems was related to an increased milk yield per cow, so that fewer cows were needed to produce the same amount of milk, and to an increase in the replacement rate due to increased milk yield. However, this reduction would be smaller if male calves from the specialised dairy herd were fattened to produce bulls instead of being consumed as veal.

The effect of this specialisation on environmental impacts remains to be analysed. In general, specialised dairy production reduced the impact per unit of milk produced but increased the impacts related to beef due to the increase in beef production from the suckler herd to compensate reduced beef output from the dairy herd. For example in Sweden, the increase in milk production from 6.1 to 8.2 t energy corrected milk (ECM)/cow/year from 1990 to 2005 reduced GHG emissions from 1.27 to 1.02 kg CO_2/kg ECM (Cederberg et al., 2012). The GHG emissions from average Swedish beef production, contrary to milk, increased from 18.0 to 19.8 kg CO_2 per kg carcass weight from 1990 to 2005 (Cederberg et al., 2012). Martin and Seeland (1999) reported that specialisation in cattle production slightly increased the emissions per unit of protein produced if the milk and beef production were kept constant. In this example, milk yield increased from 5.5 to 7.1 t/cow/year after two selection cycles and the number of dairy cows therefore decreased by 22.4% while the number of beef cows increased by 18.9% to produce the same amount of milk and beef. In the same way, Zehetmeier et al. (2012) evaluated the effects of using either dual-purpose cows producing 6.0 or 8.0 t milk /cow/year or Holstein-Friesian cow producing 10.0 t/cow/year to produce the same amount of milk and beef as the system with cows at 6.0 t milk/year. These authors reported that the increase in milk yield from 6.0 to 8.0 t/cow/year did not affect the overall GHG emissions but further increase in milk yield to 10.0 t /cow/year increased GHG emissions by 8% to produce the same amount of milk and beef.

In this thesis, the effect of specialisation has been analysed in Chapter 4 through the comparison between specialised Holstein and dual-purpose Normande dairy systems. Overall, the Holstein dairy systems (including beef from a suckler system) had 7-17% lower impacts (for climate change, energy use and acidification) than the Normande dairy system to produce the same amount of milk and meat, except for EP and LO impacts, which were less than 5% higher. This can be explained first by Normande cows having a low milk yield (5.7 t/cow/year) relative to Holstein cows (from 6.7 to 8.7 t/cow/year), so to produce the same quantity of milk, the Normande dairy system required more cows and heifers in the herd. Second, for the Normande system, all calves were kept and raised as heifers used for replacement and heifers not used for replacement were sold before the first calving (at 3 years age), whereas they were principally sold as 2-week calves for the Holstein systems. Third, the Normande dairy system is based on grass only. To ensure the supply of grass for the Normande dairy herd, the rate of N fertiliser application per ha of grassland was high. In contrast, in the Holstein systems studied here, an increased proportion of silage maize in the forage area was associated with reduced N fertiliser use per ha of grassland. In addition, in the studied system silage maize required a lower N fertiliser rate and had a higher DM yield than grassland.

This thesis only assessed some particular production systems for a dual-purpose breed and a specialised breed; these should not be considered as representative for dual-purpose and specialised French dairy systems. The effects of specialisation should be assessed at different levels (national, regional or global level) and should be connected with milk and beef demand at the same level. It should be noted that the effects of specialisation on e nvironmental impacts depend much on assumptions regarding the production of "missing" beef. It depends in particular on how suckler-beef is assumed to be produced and how its impacts are assessed.

To add a different perspective, I think that, from a global point of view, milk production should be considered as a priority means of providing animal protein to the increasing world population. In this respect, the necessity to keep beef production at its present level can be questioned. Indeed, the "missing" beef could be partially compensated for by other types of meat such as pork and poultry. Finally, dairy herds can produce more beef if male calves were fattened to produce bulls instead of calves. Therefore, efforts that aim to change consumption patterns may contribute to answering the question whether specialised dairy production contributes to reduce impacts of cattle production.

Intensification has been defined as producing more of the desired products per unit area of land (Menzi et al., 2010; Smith et al., 2010). Dairy and suckler cattle production systems have been intensified through increased N fertilisation of grassland and/or increasing animal productivity by the use of maize silage and concentrate feed. For both dairy and suckler cattle production systems intensification through higher N application rates allowed increased stocking rates, but did not reduce GHG emissions per unit of product (Haas et al., 2001; Casey and Holden, 2006; Basset-Mens et al., 2009; White et al., 2010; Clarke et al., 2012). The latter can be explained by the fact that the increase in N fertiliser was higher than the increase in animal productivity (Crosson et al., 2011), which led to higher emissions of nitrogenous pollutants per unit of animal product. In contrast, the increase in animal productivity due to increased use of maize silage and concentrate feed reduced GHG emissions per unit of animal product for both dairy and suckler-beef cattle production systems (Basset-Mens et al., 2009; White et al., 2010). Dollé et al. (2011), using a sample of 153 dairy farms in France, reported a modest decrease in GHG emissions per litre of milk with increasing cow productivity.

In Chapter 4 of this thesis, intensification consisted of increasing milk yield per cow for Holstein dairy systems by using more maize silage and concentrate feed at the expense of grass and grass-based forages. These production systems produced the same amount of milk but less beef with higher milk yield/cow/year. Independent of the co-product handling method used, impacts decreased both per kg of milk and per kg live weight of animals with increasing intensification, except for the eutrophication impact and when co-products were handled by system expansion. One of the most effective impact mitigation strategies for the suckler-beef system was decreasing the age of the first calving from 3 to 2 years (Chapter 3). This strategy can be considered as intensification mainly based on herd management, resulting in a major reduction of grass and grass-based forages used for the herd with the same level of animal output.

A study of the U.S. dairy industry showed that increasing animal productivity can significantly reduce the national herd size, which substantially reduced GHG emissions of the sector (Capper et al., 2009). However, this study did not assess the effect on the beef production sector. As discussed, it cannot be concluded that increasing animal productivity is an effective GHG mitigation strategy for the dairy sector when its effect on the beef sector has not been considered. Moreover, further studies are needed to investigate how the side-effects of increasing milk yield per cow such as loss of fertility, increased replacement rate, and other health problems can counter mitigation effects. The environmentally optimum level of milk yield per cow while considering indirect effects on the beef sector and other side effects still remains to be determined.

Overall, intensification can reduce impacts per kg product when the increase in productivity is larger than the increase in impacts associated with additional inputs used (N fertiliser, feed). However, intensification often increases impacts per unit of land, because less land is occupied to produce the same output. The later has been clearly demonstrated by the application of decreasing calving age in the suckler-beef system (Chapter 3) and by the comparison of Holstein dairy systems with different milk yield levels (Chapter 4). For dairy systems, the area becoming available due to increasing milk yield per cow can be used to produce annual crops because it is arable land. As for suckler-beef systems, the area becoming available was mainly permanent grassland. The alternative land use as forest can be considered to have a short- to mid-term influence on GHG emissions by sequestering C in biomass. However, reintroducing forest into regions dominated by grassland-based farming systems may not be welcomed by all stakeholders. Nonetheless, forest can be introduced in cattle systems via silvopastoralism, which is generally practised in developing countries but less commonly in developed countries. Similarly, on-farm area can be redesigned, for example by increasing the area covered by hedgerows or planting forests on fields that livestock have difficulty accessing.

The importance of world agricultural intensification on GHG mitigation was highlighted by Burney et al. (2010), who estimated the net effect of historical agricultural intensification between 1961 and 2005 on GHG emissions. They suggest that emissions avoided due to avoided land-use changes (i.e. cropland expansion) through agricultural intensification would be larger than the emissions due to increased input use. The net effect of world agricultural intensification to meet global demand for food has avoided emissions of up to 590 gigatons (Gt) of CO_2 eq. since 1961 (Burney et al., 2010). This trend in particular concerned livestock production because it uses agricultural land to feed animals.

Gerber et al. (2011) reported a significant relationship between milk production per cow and GHG emissions per kg of milk across countries and regions and suggested the increase in productivity as a viable GHG mitigation approach, in particular for the areas, often in developing countries, where milk yield per cow is lower than 2000 kg/year. However, Gerber et al. (2011) did not account for the effect of increasing milk yield on beef production. Increased milk yield for smallholder dairy production systems in East Africa and South Asia can be obtained through improvements in animal genetics, availability of quality feed and animal health (McDermott et al., 2010). There is a growing consensus that more research should be focused on intensification of agricultural production by improving management and efficiency of inputs based on knowledge of natural processes in agro-ecosystems, which could be defined as sustainable intensification, in order to improve synergy between production and environment (Tilman et al., 2011; Garnett and Godfray, 2012).

6.3. Accounting effects of LU and LUC on dynamics of soil C in a LCA approach

Soil C sequestration is the mechanism that contributes most (up to 89%) to GHG mitigation of agricultural activities (IPCC, 2007). However, soil C sequestration is both reversible and vulnerable to disturbance due to direct human-induced effects (i.e. changes in management practices (LU) and LUC), and indirect human-induced and natural effects (Smith, 2005). When direct and indirect effects of LUC are taken into account, the potential of biofuels to reduce GHG emissions relative to fossil fuels is questionable (Searchinger et al., 2008; Laborde 2011; Marelli et al., 2011). The LUC from the current use (e.g. food or feed crops) into use for another purpose (e.g. grassland) is defined as direct LUC and the resulting effects are called direct LUC effects (Overmars et al., 2011). As a consequence, the original crop production will have to be realised elsewhere (through intensification, expansion and displacement) and/or by a change in consumption (due to increasing in crop prices), these are called indirect LUC effects (Overmars et al., 2011). Considering the effects of LULUC is a critical issue in the environmental assessment of agriculture; both in livestock (FAO, 2010; Flysjö et al., 2012; Weiss and Leip, 2012) and crop production (Kløverpris et al., 2008; 2010).

The ILCD guidelines (JRC, 2010) recommend that LCA studies must be based on the most recent data and emission factors according to IPCC. In the IPCC methodology only direct human-induced effects can be considered (Smith, 2005). Regarding dynamics of soil C, the activities considered as direct human-induced effects under the Kyoto Protocol are afforestation, reforestation, deforestation, forest management, cropland management, grazing land management and revegetation (Smith, 2005). Changes in soil carbon stocks may result from indirect human-induced and natural effects such as the increasing concentration of CO_2 in the atmosphere, increasing nitrogen deposition due to global N enrichment and climate change (e.g. increased temperature, precipitation, extreme events) (Smith, 2005). It has also been highlighted that indirect effects may significantly affect the permanence of human-induced soil C sinks and it is a major challenge to identify the contribution of direct human-induced effects, indirect human-induced effects and natural effects when a change in soil C stocks is observed (Smith, 2005; Soussana et al., 2010).

Effects of LU

In most studies (Phetteplace et al., 2001; van der Werf et al., 2009; O'Brien et al., 2012), as well as in the IPCC guidelines (IPCC, 2006), soil C stock for permanent grassland is considered to be stable. However, based on measurements using the eddy covariance technique Soussana et al. (2007 and 2010) suggest that permanent grassland is a net sink for atmospheric CO_2 in soil. Literature shows a high variability of C sequestration by grassland in different countries and regions. Pelletier et al. (2010) assume that C sequestration rates for improved and unmanaged pastures in the USA are about 0.12 and 0.40 t C/ha/year, respectively. Leip et al. (2010), for a European expertise, estimated C sequestration for managed permanent grassland at 0.24 t C/ha/year. Schils et al. (2005) estimate C sequestration rates at 1.77 t and 1.36 t C/ha/year for grass and grass/clover systems, respectively. In a sensitivity analysis, O'Brien et al. (2012) assume that Irish permanent grassland could sequester 2.5 t C/ha/year, according to Soussana et al. (2010). In France, Gac et al. (2010), Dollé et al. (2011) and Veysset et al. (2011) propose that permanent grassland sequesters 0.2 to 0.5 t C/ha/year, depending on the age of grassland, according to data by Arrouays et al. (2002). In addition, Gac et al. (2010) and Dollé et al. (2011) propose to attribute C sequestration in hedges present on the farm to the farm's products; sequestration rate used by these authors is 1.25 t C/1000 linear meter of hedge.

So literature values for storage rates for soils under permanent grassland vary from 0.1 to 2.5 t C/ha/year. Including C sequestration in soil under permanent grassland in an LCA study is not simple: 1) values proposed for rate of sequestration are extremely variable (due to soil properties, management practices, climatic conditions (Conant et al., 2001; Klumpp et al., 2011),and to methods used to assess soil C stock changes, which often may not have been captured in full because soils have not been sampled deep enough (Leifeld et al., 2011) 2) determining the direct human-induced part of these values is very challenging (Smith, 2005); 3) in some cases soil organic C can be saturated due to climate or management (Angers et al., 2012).

Facing the high variability of data for C sequestration in grassland in the literature and the difficulty to determine the direct human-induced part for this sequestration, in this thesis (Chapters 2, 3 and 4), we applied a modest C sequestration rate for grassland in France. We assumed 0.2 t C/ha/year for permanent (i.e. more than 30 years old) grassland, based on data by Arrouays et al. (2002). In Chapter 5, we used the Roth C model (Coleman and Jenkinson,

1996) to simulate, among others, the effects on soil C dynamics of permanent grassland and of the conversion of temporary into permanent grassland. In this chapter, we investigated a significant change in on-farm cropping pattern (consequences of changing a maize-silage-based to a g rass-based dairy farm) which induced substantial changes in above- and belowground biomass. In order to assess the effects of these complex changes on on-farm soil C dynamics, the use of the Roth C simulation model seemed the most appropriate approach. For example, permanent grassland and the conversion from temporary to permanent grassland were estimated to sequester 0.29 and 0.46 t C/ha/year, respectively. However, the effects of changes in management practices on soil C sequestration occur when these practices will be maintained over a long period. Otherwise, the stored C may be lost even before it has reached a new equilibrium (Arrouays et al., 2002).

Effects of LUC

Estimating the effects of LUC (changes in area and emissions associated) and attributing the associated effects on soil C dynamics to products and an amortization period is also a critical issue, because it is difficult to determine the drivers of LUC (Cederberg et al., 2011; Flysjö et al., 2012). In this thesis, we adopted the method proposed by Prudencio da Silva et al. (2010) to estimate direct LUC related to soybean in Brazil. This method estimated, for the 2005-2008 period, that annually 1% of the soybean produced in Central-West Brazil comes from tropical forest and 3.4% from shrubland, whereas soybean from South Brazil does not contribute to LUC. To conform better to current practice regarding the effect of LUC on C emissions due to conversion of Brazilian forest to cropland, we decided to adopt a value of 740 t CO_2/ha, as recommended in PAS 2050 (2008) among others, instead of the value of 120 t CO_2/ha used in the ecoinvent database (Jungbluth et al., 2007). The latter estimate corresponds to the estimated of 20% of the above-ground biomass which is burnt, but the remaining 80% is ignored, the reason for this is not specified (Prudêncio da Silva, 2011).

As for LUC (both direct and indirect) due to the increase in demand for crop products (Chapter 5), we used GTAP and LEITAP simulation results as reported by Edwards et al. (2010). This type of worldwide economic model to predict global LUC caused by an increase in demand for feedstock has been frequently used for biofuel studies (Searchinger et al., 2008; Kløverpris et al., 2008; Hertel et al., 2010; Edwards et al., 2010). However, model outcomes have high variability and uncertainty due to model characteristics, their preference setup, hypotheses and assumptions (Prins et al., 2010; Laborde, 2011; Marelli et al., 2011).

Data for soil C dynamics (emissions and sequestration) also are highly uncertain due to variability in measurements (Soussana et al., 2010), lack of data for specific areas, in particular developing countries, and lack of knowledge of the processes involved in C cycling (Dawson and Smith, 2007). The default values of IPCC Tier 1 method (IPCC, 2006) have usually been used due to the lack of adapted data.

Audsley et al. (2009) have used a "top-down" approach and argued that all land occupation contributes to LUC. These authors propose to attribute total LUC emissions due to agricultural activities to total commercial agricultural land use, which resulted in a LUC-factor of 1.4 t CO_2 per ha of agricultural land. The advantage of this method is that it strongly emphasises the need to efficiently use agricultural land, i.e. the increase in efficiency of all types of land occupation can reduce LUC. However, it does not help to identify the real cause or the main drivers of LUC (e.g. increase in biofuel demand, increase in food demand) and then to identify specific mitigation measures.

The time taken for soil C accumulation to reach a new equilibrium after a change in management practice or LUC is highly variable, from less than 10 years in the tropics to up to 100 years in temperate regions (Smith, 2005). In addition, soil C changes occur most rapidly soon after the implementation of LULUC, and then at a lower rate until soil C reaches a new equilibrium (Arrouays et al., 2002; Smith, 2005). Arrouays et al. (2002) report that C storage occurs mainly during the first 30-40 years after a conversion from cropland to grassland, and then slows down. When grassland is converted to cropland results in C losses are more rapid and occur mainly during the first 20 years.

Therefore, the amortization period for considering C sequestration or release is an important factor which can affect the balance of GHG emissions in an LCA approach. The amortization period is generally 20 years according to European legislation and IPCC guidelines (Marelli et al., 2011) or 30 years according to US legislation (Marelli et al., 2011), as applied in Searchinger et al. (2008) and in the GTAP model (Edwards et al., 2010). The amortization period is supposed to correspond to the period for which dynamics of soil C are strongest. However, Audsley et al. (2009) propose that no amortization period is needed due to the fact that the current demand drives the current LUC. In this thesis (Chapter 5), we applied a 30 years amortization period for LUC according to the GTAP model.

Considering LULUC or not, as well as the approach chosen to consider LULUC, substantially affects LCA results, as shown by the values for CC and CC/LULUC in this thesis, especially in Chapter 5. This is also in line with Flysjö et al. (2012), who compared several approaches to account for LUC for soybean meal. Therefore, it is important to report results, both with and without LULUC, and to emphasize the approach used and hypotheses, as well as the sources of data (Flysjö et al., 2012). More generally, to better account for the effects of LULUC, it is necessary to have a consensual methodology and to harmonise the critical aspects of LULUC emissions such as defining the driver of LULUC, the time period for which LULUC is considered (e.g. the latest 20 years), the use of simulation models or static factors for both changes in area and emission factors, and the amortization period.

Generally extensive (or organic) agricultural production systems have lower impacts per ha, but require more land to produce the same quantity as more intensive systems (Tuomisto et al., 2012). However, land (especially land suitable for agriculture) is a limited resource, and if agricultural land were used more efficiently, the pressure on conversion of natural land (forest) or pasture to crop land would be reduced. This not only makes sense in term of avoided CO_2 emissions due to conversion from forest to pasture and cropland, or pasture to cropland, but also in term of preservation of biodiversity for the global ecosystems. This illustrates an asset of the LCA approach, which allows the analysis of production systems by using different functional units and several environmental impact categories. For example, if we only investigate the effect on climate change, one does not observe the effect on land occupation and its consequences on LUC. Moreover, the increase in demand, and subsequent increased prices of food reduce access to food, in particular for low-income households and countries. Therefore, the effects of LUC, not only on GHG emissions, but also on biodiversity, affordability of food and hence quality of life need to be further investigated.

6.4. CLCA and co-product handling methods

6.4.1. CLCA approach

Consequential LCA aims to describe how physical flows (i.e. resources, material, emissions, etc.) may change as a consequence of a decision, while Attributional LCA accounts for immediate physical flows involved across the life cycle of a product. The modelling principles are the same for ALCA and CLCA, what primarily distinguishes CLCA from

ALCA is the use of, respectively, affected processes instead of average processes (Zamagni et al., 2012). Direct comparison between ALCA and CLCA outcomes does not make sense, because the two approaches aim to answer different questions.

The application of a correct LCA method is strictly related to the type of question (purpose) and is crucial to providing relevant answers (Zamagni et al., 2012). Nevertheless, most LCA practitioners choose one methodology independently of their research question (Thomassen et al., 2008). In this thesis, the aim of Chapter 5 was to examine the possible consequences in terms of environmental impacts of the conversion of maize-silage-based dairy production to grass-based dairy production. In order to answer this question, the CLCA method was implemented. However, had the question been the comparison of the environmental impacts of maize-silage-based and grass-based dairy production, then the application of ALCA as in Chapter 4 would have been more appropriate.

In Chapter 3, we applied the ALCA approach to examine the effects of mitigation practices on the environmental impacts of a suckler-beef production system. In fact, this is a comparison of several farming practices with the baseline scenario, while producing the same amount of animal product to identify which is “the best farming practice” in terms of environmental impacts. After this step, if suckler-beef farmers apply the practices which reduce land requirement per unit of product (e.g. reduction in grass losses, calving age at 2 years old or simultaneous mitigation practices) and increase their production capacity, then more beef will be produced. This raises the question of the environmental consequences of this increase in suckler-beef production, and the application of CLCA would be appropriate. These increases in production (marginal production on released land) through higher efficiency in land use will require more inputs (mineral fertilisers, meals) and cereals for the additional herd, but also produce more beef. This additional beef can avoid production elsewhere, for example, in Brazil on deforested area. It will be interesting to explore these situations with CLCA by using several scenarios. In addition, it would be prudent to examine possible negative indirect effects before applying the proposed mitigation practices. This may help to avoid proposing solutions which, on closer examination present major drawbacks, like the case of first generation biofuels, which cause substantial GHG emissions due to indirect effects on LUC (Searchinger et al., 2008).

CLCA is a sophisticated modelling technique (Zamagni et al., 2012) and it is not easy to determine and evaluate the possible consequences in the future of an action/decision (Smyth and Murphy, 2011). The chain of consequences may seem to be a never-ending story (Weidema et al., 1999; Zamagni et al., 2012). The integration of market mechanisms through economic models and the use of scenario modelling have been recommended for CLCA modelling with respect to the plausible future technology development and to support the identification of relevant mechanisms (Ekvall and Weidema, 2004; Earles and Halog, 2011; Zamagni et al., 2012). Nevertheless, most CLCA studies have implied simplification, assumptions, resulting in additional uncertainty. Thomassen et al. (2008) found that results of CLCA modelling are more uncertain than ALCA due to the inclusion of market prospects. Ekvall and Weidema (2004) stated that the future is inherently uncertain and that the actual consequences of decisions are highly uncertain. This uncertainty can be quantified by using scenario modelling with several alternative market situations to perform a sensitivity analysis (Thomassen et al., 2008; Zamagni et al., 2012).

6.4.2. Co-product handling methods

Determining the impacts of co-products is an important and much debated subject in LCA. Although many methods have been proposed, it is difficult to recommend to stakeholders or to convince consumers which method is most appropriate. Among ALCA studies, economic allocation is most frequently used (de Vries and de Boer, 2010). This may seem to be inconsistent with ISO recommendations (ISO, 2006) which privileges process subdivision and system expansion before allocation methods, and biophysical allocation rather than economic allocation. In this thesis, we followed the ISO recommendation that methods for co-product handling (allocation, system expansion) were only applied where the process could not be subdivided any more. The common process (phase) for both suckler-beef and dairy production systems included production cows, replacement heifers, new-born calves and breeding bulls for the suckler herd. The products of this phase, for which co-product handling methods were applied, were cull cows, sold (or weaned) calves which were not used for replacement, cull breeding bull (suckler herd) and milk for the dairy herd.

The application of several co-product handling methods resulted in large differences in the impact results of product outputs. In this thesis (Chapters 2 and 4), the range of variation in impacts per kg of live weight due to allocation methods (biophysical, mass, protein, or economic allocation) was smaller for different types of animals in the suckler-beef production system (-51% to +20%, depending on types of animal, systems and allocation methods) than for those in dairy production systems (-49% to +108%). Higher variation in impacts per kg live weight was due to differences in allocation methods and in rations and animal growth for the rearing period after the common phase for different types of animal outputs within a system. Apart from these differences, higher variation for impacts per kg live weight in dairy systems compared to the suckler-beef system was found because the impacts of sold animals in dairy systems depended first on those of milk according to the different allocation methods. It was shown that different types of animals in suckler-beef and dairy systems contributed differently to impacts and it is important to differentiate animal categories to identify priorities for mitigation and to identify appropriate practices for each type of animal.

The range of variation in impacts per kg of FPCM due to allocation methods was smallest (17%) compared to that per kg of live weight in both suckler-beef and dairy systems. This is in line with the results by Cederberg et al. (2003), Flysjö et al. (2011) and Kristensen et al. (2011). However, the range of variation was larger (up to 40%) when system expansion was used while assuming that animals from dairy system can be used to replace beef animals or a mix of animals for beef and pork, or beef, chicken and pork (Cederberg et al., 2003; Flysjö et al., 2011; Kristensen et al., 2011). The impact results obtained with system expansion are strictly dependant on the assumptions used. It would be more appropriate to use market information and mechanisms and to integrate economic models to identify possible substituted products and expanded systems, in the same way as possible consequences of an action/decision are identified in CLCA. Obviously no model is perfect, the use of scenario modelling is recommended to investigate the variability and uncertainty related to the choice of the expanded system.

Last but not least, LCA has been criticised for considering only negative impacts and excluding the positive aspects of agriculture (e.g. Bockstaller et al., 2010). Several approaches for assessing biodiversity of agricultural production system have recently been developed (e.g. assess the overall species diversity, potential regional mammal species extinction, etc.; de Baan et al., 2012; Nemecek et al., 2012). A methodological consensus for defining impact on biodiversity in LCA is required but not yet established.

In this thesis, we tried to integrate the positive aspects of a suckler-beef production system, which is mainly based on (permanent) grassland, such as a contribution to landscape quality, to biodiversity conservation and rural development by considering these functions as “public goods”. These “public goods” are a co-product of this type of farming system the maintenance and preservation of which are promoted by agro-environmental subsidies of the "Second Pillar" of the CAP. We do believe that LCA can consider these positive aspects simply by the application of co-product handling methods (Chapter 2).

6.5. Conclusions and perspectives

This thesis adds several original contributions to the field of environmental systems analysis of animal production systems. First, thanks to a detailed representation of beef and milk production systems it assesses the environmental performance of a wide range of management practices and feeding strategies for these systems. This research also proposes a new approach to take into account ecosystem services of cattle farming systems based on a gro-environmental subsidies for grassland. This thesis contains an in-depth exploration of the issue of LULUC by considering alternative land-use options and by addressing the substantial consequences due to a preference for grass-based milk. The latter issue was explored using a CLCA approach, providing the material for, we hope, the second paper (following Thomassen et al., 2008) implementing CLCA for cattle production.

Several conclusions can be drawn from this thesis. First, enteric CH_4 fermentation and grassland production were the main contributors to climate change (39-45%) and other impacts (27-83%, depending on i mpacts and systems) of suckler-beef production systems, respectively. The suckler cow-calf herd with finishing heifers substantially contributed (73 to 97%, depending on impacts and systems) to the environmental impacts of the whole suckler-beef production system.

Second, it is difficult to greatly reduce impacts of whole suckler-beef production systems. Feeding strategies (increased proportion of concentrate, lipids rich in omega-3 FAs) applied for the bull-fattening herd had small effects at the whole system level due to its small contribution. For the suckler cow-calf herd, the most effective mitigation practice was decreasing calving age from 3 to 2 years. Simultaneous application of several compatible practices can substantially mitigate the impacts of suckler-beef production system. Some mitigation options (reducing grass losses during grazing, fattening heifers not used for replacement and reducing calving age) resulted in reduction of land occupation for the herd without reducing animal product outputs. The introduction of forest as an alternative land-use option in beef-cattle farms seems promising for short-and mid-term GHG mitigation and merits further exploration.

Third, using the ALCA approach, enteric CH_4 fermentation and grassland production were the main contributors to climate change (45-50%) and other impacts (19-90%, depending on impacts and systems) of dairy production systems, respectively. Independent of co-product handling methods, the grass-based dairy system resulted in higher impacts per kg of milk than the maize-silage-based dairy system (except for eutrophication). Independent of co-product handling methods, increasing milk yield per cow by increasing energy intake and associated more intensive management (first calving at 2 years) decreased the impacts of milk and beef, respectively. The impacts of milk produced with the Normande dairy system were higher than those with Holstein dairy systems. Nevertheless, the former produced more beef than the latter, with the same quantity of milk. If beef from a suckler-beef production system was used to compensate for "missing beef" in Holstein systems relative to the Normande system, impact differences between these systems were smaller.

Fourth, the environmental consequences of the conversion from a maize-silage-based to a grass-based dairy system were assessed using the CLCA approach. Respectively, climate change and land occupation impacts of a grass-based system (to produce the same quantity of milk, beef and cash crops) were higher than those of a maize-silage-based system by 7 and 9% with the estimation of GTAP model and by 51 and 14% with LEITAP model (expressed for the whole system). As meeting increased preference for grass-based milk is the driver of farm conversion, the climate change and land occupation impacts of grass-based milk were higher than those of maize-silage-based milk by 13 and 82% with GTAP and by 97 and 123% with LEITAP, respectively (expressed per kg of milk). The conversion from a maize-silage-based to a grass-based dairy system will have substantial consequences on LUC outside the farm. Thus, it is important to assess the consequences of changes in a production system on the processes beyond the system boundary, particularly on LUC, by using the CLCA approach. The integration of global economic models assessing the effects on LUC is necessary in CLCA modelling.

Fifth, the application of several co-product handling methods for suckler-beef and dairy production systems showed large differences in impacts of animal product outputs. Both for suckler-beef and dairy systems, the allocation based on protein content and economic values resulted in smaller differences in impacts of animal products compared to other co-product handling methods (mass and biophysical allocation, and system expansion).

Finally, the enrichment of ruminant diets with omega-3 fatty acids allows the improvement of product quality without substantially affecting the environmental impacts of suckler-beef and dairy production systems.

These conclusions address the main objectives of this thesis, but complementary analyses may be performed. In addition, results show some limits of the LCA approach as it is developed now, and methodological and conceptual improvements are needed. Some clear perspectives can be drawn.

The most frequent French production systems for suckler-beef and dairy have been studied. For suckler-beef production systems, the possibility for modifying management practices and feeding strategies for the suckler cow-calf herd is modest due to its low-input and low-cost nature. In contrast, several management and feeding practices allow the reduction of impacts for the bull-fattening herd. Even though the overall reduction for the whole system is modest, it can contribute to reduce impacts for both farms specialised in fattening bulls from the suckler herd as well as for farms fattening bulls from the dairy herd.

For dairy production systems, it is important to assess the effects of the principal mitigation practices, in particular for feeding strategies to reduce enteric CH_4 emissions. This thesis only evaluated the effects of omega-3 FA from linseeds. However, other sources of lipids (e.g. rapeseed) and additives may potentially reduce enteric CH_4 emissions. It is necessary to evaluate the effects of these strategies for total GHG emissions and other environmental impacts at the whole system scale. In addition, potentially additional impact reduction for dairy production system can be obtained from management practices, especially for grassland (e.g. reduction of N fertiliser, reduction of grass losses during grazing).

One major issue is how to account for C sequestration by permanent grassland. This is the subject of much debate. Although, according to LCA principles only direct human-induced effect should be taken into account, we wonder whether total C sequestration might be included in calculations, for a better assessment of the climate change impact of production systems. However, this probably would not be considered a good idea by those who are confronted (now or in future years) with permanent grassland in which net release of carbon rather than sequestration occurs. It is likely that improvements in knowledge and new models will appear in the coming years, considering soil, climate and management. For the moment, we suggest that conducting sensitivity analyses for C sequestration may help to quantify the uncertainty associated with this phenomenon.

Research presented in this thesis evidenced the importance of studying the consequences of land use changes. In Chapter 3, forest has been presented as an alternative land-use for “released” permanent grassland in suckler-beef systems. Further research is required to better explore the environmental and economic strengths and the limits of this change. Additional knowledge is required on the optimisation of forest management, and on the amortization over time of land use change.

Further methodology development must be focused on the CLCA approach by integrating market mechanisms and sophisticated global economic models. It is necessary to compare several economic models and use scenarios while defining affected processes and indirect consequences of a change in the production system. It is especially important to better account for LUC outside beyond the system boundary. However, the implementation of CLCA is complementary with ALCA and the choice of LCA approach must be justified with the purpose of study.

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Annex

Life cycle assessment of three bull-fattening systems: effect of impact categories on ranking

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MODELLING ANIMAL SYSTEMS RESEARCH PAPER

Life cycle assessment of three bull-fattening systems: effect of impact categories on ranking

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SUMMARY

The aim of the current study was to analyse the environmental impacts of fattening bull systems using life cycle assessment (LCA). Three contrasting bull-fattening systems practised in France were compared. Diets H, MS and C differed by the nature of the forage consumed (hay, maize silage and wheat straw, respectively) and the proportion of concentrate, i.e. ground maize grain and soybean meal (0·51, 0·37 and 0·86, respectively) in the diets. Diet MS resulted in the lowest cumulative energy demand and in the highest acidification potential per kg of body weight gain (BWG). Eutrophication potential per kg of BWG was highest for diet C and the lowest for diet H. The relative contribution of eutrophication and acidification impacts by feeds and manure varied according to diet. The system using a hay-based diet resulted in the highest land occupation per kg of BWG and in the lowest impact per ha of land occupied for all impacts. It was found that the use of LCA, involving a multi-criteria assessment allowing the expression of results according to several functional units (kg of BWG and ha of land occupied) is essential to analyse the effectiveness of a pollution reduction strategy.

INTRODUCTION

Environmental impacts have become a major issue for animal production, especially for meat production. Among the impacts, greenhouse gas (GHG) emissions are now considered as the main concern due to the dramatic consequences for climate change (Thomas *et al*. 2010). However, other impacts such as energy demand, eutrophication, acidification and impacts associated with land use are of major importance, depending on the country (Steinfeld *et al*. 2010). At the same time, meat consumption is expected to increase in the world in the future, due to demographic pressure and economic growth. This affects mainly pork and poultry, but beef is expected to increase too (FAO 2009). Therefore, it is important to reduce the environmental impacts related to the beef sector. Although more than 0·70 of environmental impacts of beef production systems, when expressed by kg of product, arise from the cow–calf phase (Pelletier *et al*. 2010; Nguyen *et al*. 2012), it is important to investigate strategies for decreasing impacts during the fattening phase. For example, in France, beef farms specialize in either the suckler cow–calf phase based on forage feeding or in the fattening phase based on diets with higher amounts of concentrates. Farms specialized in fattening require mitigation strategies to decrease emissions of pollutants. Although very little information is available in the literature about the effect of the type of diet on environmental impacts, it is hypothesized that the choice of feeding system may modulate the extent of air and water pollution, of resource consumption and of land occupation. The current study aims to assess environmental impacts by life cycle assessment (LCA) in three contrasting systems representative of different feeding systems of bull-fattening in France. The impact of these finishing systems on GHG emissions have been described in Doreau *et al*. (2011). Large differences between feeding systems have been shown. The current paper analyses the implications for eutrophication, acidification, cumulative energy demand and land occupation, per kg of product and per ha of surface.

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MATERIALS AND METHODS

System characteristics

The three systems correspond to typical French bull-fattening farms and differ in the nature of the forage consumed and the proportion of concentrate in fattening diets, and by the region of France where these systems are most likely to be found. On a dry matter (DM) basis, diet H consisted of 0·49 natural grassland hay, 0·41 ground maize grain and 0·10 soybean meal; diet MS consisted of 0·63 maize silage, 0·21 ground maize grain, 0·16 soybean meal; diet C consisted of 0·70 ground maize grain, 0·16 soybean meal and 0·14 wheat straw. Diets H, MS and C were representative of diets fed to finishing bulls in the Centre of France (Auvergne), the North West of France (Brittany) and the South West of France (Aquitaine), respectively. The fattening period occur over 175, 147 and 131 days for H, MS and C, respectively. The main features of the three bull fattening processes were determined during a feeding trial and are reported in Table 1. The boundaries of the fattening systems were limited to the production and delivery of the components of the diets (natural grassland hay, maize silage, wheat straw, maize grain and soybean meal), the production and delivery of inputs used to produce these components (e.g. seed, diesel, tractors and fertilizers), associated upstream processes and emissions from manure management. The application of the manure for the production of these components was included. Buildings and veterinary medicines were not included because of lack of data. The transport and slaughter of animals leaving the system were also not included.

Table 1. *Performances of fattened animals in three bull-fattening systems*

	Diet*		
	H	MS	C
Number of days of fattening†	175	147	131
Feed intake (kg DM/d)‡	6·74	6·75	6·26
Average BWG (kg/d)†	1·49	1·71	1·86

* Diet H = 0·49 natural grassland hay + 0·41 maize grain + 0·10 soybean meal; Diet MS = 0·63 maize silage + 0·21 maize grain + 0·16 soybean meal; Diet C = 0·14 wheat straw + 0·70 maize grain + 0·16 soybean meal.
† Mialon *et al.* (2008).
‡ Doreau *et al.* (2011).

Emissions from feed production and animals

Grassland practices for hay production were based on data provided by the French extension service (Institut de l'Elevage 2008). The crop production inputs and management practices used were based on French government statistics (AGRESTE 2006); yield levels were the averages for 2004–7. Slurry produced by bulls in the fattening system was stored without natural crust cover and its emissions in housing and in storage were considered to be part of the animal production system and accounted for. After storage, resource use and emissions associated with the transport and application of this slurry, and of organic fertilizer in general, were considered to be part of the crop production system. Maize grain and maize silage grown in Brittany were mainly fertilized with pig slurry, while maize grain and wheat in other regions received mainly mineral fertilizer. Estimations of nitrate-N emitted from crop production was based on Basset-Mens *et al.* (2007), considering the nature of the crop and the duration of the subsequent period without the presence of a crop. For grassland production, nitrate-N emitted was based on Vertès *et al.* (1997), considering the management practices and harvest methods. Data for fertilizer use, yield level and nitrate-N emitted of feed ingredients are summarized in Table 2. Data concerning resource use and emissions associated with the production and delivery of several inputs for crop production (fertilizers, pesticides, tractor fuel and agricultural machinery) were from Nemecek & Kägi (2007). Equations and emission factors used to calculate field-level emissions from crop and forage production and from slurry produced by bulls are presented in Table 3. It was assumed that soybean was produced in Brazil, since 0·60 of soybean meal imported in France comes from Brazil (ISTA 2009). Impacts associated with soybean production, including CO_2 emission associated with land use change from forest to soybean, were estimated according to Prudêncio da Silva *et al.* (2010) and represent the average Brazilian production, i.e. both traditional production areas and recently deforested areas.

For the processes of transformation of crop products into feed ingredients, data were based on Nguyen *et al.* (2011) for drying of maize, and Nemecek & Kägi (2007) and Jungbluth *et al.* (2007) for the production of soybean meal. Soybean transformation yields two co-products: meal and oil. Resource use and emissions for these products were allocated according to the

Table 2. *Fertilizer use, yield level and nitrate-N emitted (per ha) for the major crops serving as feed-ingredients for bull-fattening diets*

Crop	N mineral (kg/ha)	N manure (kg/ha)	P_2O_5 (triple superphosphate) (kg/ha)	K_2O (potassium oxide) (kg/ha)	Yield (DM) (kg/ha)	Nitrate-N leached (kg/ha)
Grassland hay (diet H*)	55	32	30	55	2700	20
Wheat straw (diet C†)	165	10	26	24	6010	40
Maize silage (diet MS‡)	32	210	29	0	12 400	40
Maize grain (diet H*)	169	29	48	43	7440	70
Maize grain (diet C†)	189	46	67	85	7500	70
Maize grain (diet MS‡)	32	210	29	0	6860	40
Soybean (all diets)	5·5	1·3	80	80	2708	18

* Diet H = 0·49 natural grassland hay + 0·41 maize grain + 0·10 soybean meal.
† Diet C = 0·14 wheat straw + 0·70 maize grain + 0·16 soybean meal.
‡ Diet MS = 0·63 maize silage + 0·21 maize grain + 0·16 soybean meal.

Table 3. *Published values for emissions sources, equations or emission factors used*

Pollutant/source	Equation/emission factors	References
Manure management: slurry without natural crust cover		
NH_3		
In housing	= 0·12 × N excreted (kg) × 17/14	Payraudeau *et al.* (2007)
In storage	= 0·06 × N remaining (kg)×17/14	Payraudeau *et al.* (2007)
Crop and forage production		
NH_3	= (0·02 × mineral N (kg) + 0·08 × liquid N (kg) + 0·14 × pig slurry N (kg) + 0·076 × cattle manure N (kg))17/14	Nemecek & Kägi (2007) and Payraudeau *et al.* (2007)
NO_3	See values in Table 2	Basset-Mens *et al.* (2007) and Vertès *et al.* (1997)
P leaching		
Cropping	= 0·07 kg P/(ha × yr)	Nemecek & Kägi (2007)
Grassland	= 0·06 kg P/(ha × yr)	Nemecek & Kägi (2007)
P run-off	= P run-off lost × (1 + 0·2/80 × mineral P_2O_5 (kg) + 0·4/80 × manure P_2O_5 (kg))	Nemecek & Kägi (2007)
Cropping	P run-off lost = 0·175 kg P/(ha × yr)	Nemecek & Kägi (2007)
Grassland	P run-off lost = 0·15 kg P/(ha × yr)	Nemecek & Kägi (2007)
P erosion	= 10 000 × (80 × 0·033 × 0·38 × 0·65 × effect of the vegetation cover factor) × 0·00095 × 1·86 × 0·2 kg P/(ha × yr)	Nemecek *et al.* (2003)

economic value of the products and were calculated using extraction rates (the proportion of processed products obtained from the parent product) and costs. Extraction rates were taken from FAO (2002) and costs were averages for 2004–7 from ISTA (2009).

For diet H, hay is produced on farm, maize grain is transported over 70 km by truck and soybean meal is transported by truck from sea port of Bordeaux (510 km). For diet MS, maize silage is produced on farm, maize grain is transported over 80 km by truck and soybean meal is transported by truck from Brest (110 km). For diet C, both straw and maize grains are transported by truck over 50 and 10 km, respectively; soybean meal is transported by truck from Bordeaux (310 km).

Environmental impact assessment

Environmental impacts associated with the three diets were evaluated using an LCA model, inputs and outputs being interpreted in terms of environmental impacts. The impact categories considered were cumulative energy demand, eutrophication, acidification and land occupation. The indicator result for each impact category was determined by multiplying the aggregated resources used and the aggregated emissions of each individual substance with a characterization factor for each impact category to which it may potentially contribute. Characterization factors were substance-specific, quantitative representations of the additional environmental pressure per unit emission of a substance. Cumulative energy demand was calculated according to Frischknecht *et al*. (2007) and took into account the renewable and non-renewable resources by using the conversion efficiencies of primary energy carriers. Eutrophication covered all potential impacts of high environmental levels of macronutrients, in particular N and P. Eutrophication potential (EP) was calculated using the generic EP factors in kg PO_4 equivalent (equiv.), *viz*., NH_3: 0·35, NO_3: 0·1, NO_2: 0·13, NO_x: 0·13, PO_4: 1 (Guinée *et al*. 2002). Acidifying pollutants had a wide variety of impacts on soil, groundwater, surface water, biological organisms, ecosystems and materials (buildings). Acidification potential (AP) was calculated using the average European AP factors in kg SO_2 equiv., *viz*., NH_3: 1·6, NO_2: 0·5, NO_x: 0·5, SO_2: 1·2 (Guinée *et al*. 2002). Land occupation referred to the loss of land as a resource in the sense of being temporarily unavailable for other purposes due to crop and paddocks. Land occupation is a surface multiplied by the proportion of time of use in the year, expressed as $m^2 \times yr$. Environmental impacts have been expressed per kg of body weight gain (BWG) obtained in the fattening period and per ha of land occupied.

RESULTS

The environmental impacts of maize silage for diet MS (per kg of DM) delivered at farm were lower than the impacts of maize grain for all diets, except for AP of maize grain H (Table 4), principally due to higher DM yield/ha of maize silage. Cumulative energy demand/kg DM of maize grain for diet MS was lower than those of maize grain for diet H and C; the opposite was true for AP and land occupation. There were small differences for cumulative energy demand and AP between soybean meal for H, MS and C due to the difference in transportation distances in France.

Cumulative energy demand/kg of BWG was the lowest for diet MS and the highest for diet C (18·7, 13·0 and 19·7 MJ equiv./kg of BWG for H, MS and C, respectively) (Fig. 1). Crop production was the largest consumer of energy (accounting for 0·62, 0·44 and 0·62 of the totals for H, MS and C, respectively), followed by transportation (0·15, 0·24 and 0·14 for H, MS and C, respectively) and grain drying (0·13, 0·14 and 0·12 for H, MS and C, respectively). Road transport was the main means of transportation used (0·70, 0·62 and 0·64 of cases for H, MS and C, respectively), followed by sea transport (0·25, 0·32 and 0·30 for H, MS and C, respectively). Rail transport used partly for soybean transportation in Brazil only accounted for 0·04–0·06. Impacts related to deforestation contributed for 0·04–0·08 to cumulative energy demand per kg of BWG.

AP per kg of BWG was the highest for diet MS and the lowest for diet C (31·3, 38·1 and 29·4 g SO_2 equiv./kg of BWG for H, MS and C, respectively) (Fig. 2). Ammonia from manure management was the primary contributor for AP (accounting for 0·72, 0·54 and 0·58 for H, MS and C, respectively), followed by ammonia from feed crop production (0·11, 0·33 and 0·21 for H, MS and C, respectively). Ammonia emissions from crops grown for feed for diet MS, which were twice as high as those for diet C and four times as high as those for diet H, were principally related to the application of pig slurry. Sulphur dioxide emissions accounted for 0·10, 0·07 and 0·13 of the AP of H, MS and C, respectively. The smallest contributor to AP was nitrogen oxide emissions.

EP per kg of BWG was the highest for diet C and the lowest for diet H (16·5, 19·0 and 21·5 g PO_4 equiv./kg of BWG for H, MS and C, respectively) (Fig. 2). Nitrate emissions from feed crop production were the largest contributor to EP (accounting for 0·36, 0·40 and 0·54 for H, MS and C, respectively), followed by ammonia emissions from manure (0·30, 0·24 and 0·17 for H, MS and C, respectively) and phosphate emissions from feed crops (0·25, 0·19 and 0·19 for H, MS and C, respectively). The smallest contributor to EP was

Table 4. *Environmental impacts per kg of feed ingredient (on DM basis) delivered at the bull-fattening farm*

	Cumulative energy demand MJ	AP g SO_2 equiv.	EP g PO_4 equiv.	Land occupation $m^2 \times yr$
Grassland hay (diet H*)	1·2	0·4	0·8	3·8
Wheat straw (diet C†)	0·9	0·7	1·0	0·4
Maize silage (diet MS‡)	0·8	3·3	2·6	0·8
Maize grain (diet H*)	5·2	2·6	3·6	1·4
Maize grain (diet C†)	5·1	3·5	5·8	1·5
Maize grain (diet MS‡)	3·3	6·1	4·7	1·7
Soybean meal (diet H*)	13·9	7·0	6·7	1·8
Soybean meal (diet C)	13·5	6·9	6·7	1·8
Soybean meal (diet MS‡)	13·0	6·7	6·7	1·8

* Diet H = 0·49 natural grassland hay + 0·41 maize grain + 0·10 soybean meal.
† Diet C = 0·14 wheat straw + 0·70 maize grain + 0·16 soybean meal.
‡ Diet MS = 0·63 maize silage + 0·21 maize grain + 0·16 soybean meal.

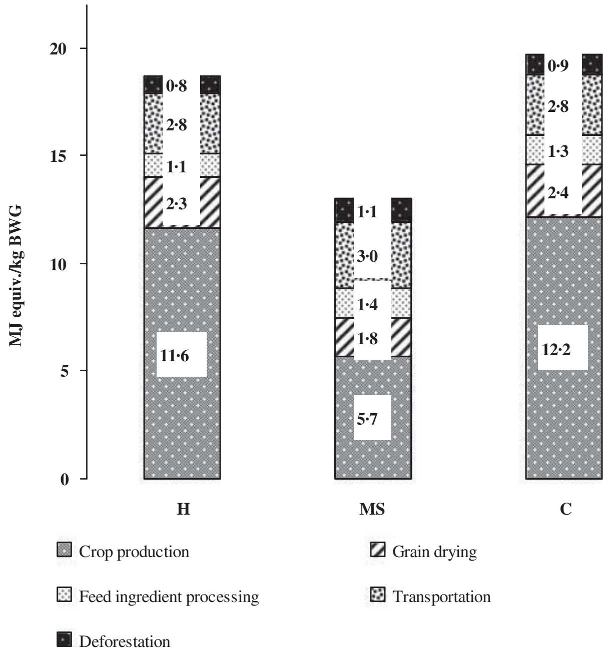

Fig. 1. Contribution of crop production, grain drying, feed ingredient processing, transportation and deforestation to cumulative energy demand for fattening systems based on hay (diet H*), maize silage (diet MS†) or concentrate (diet C‡), expressed per kg BWG. *Diet H = 0·49 natural grassland hay + 0·41 maize grain + 0·10 soybean meal. †Diet MS = 0·63 maize silage + 0·21 maize grain + 0·16 soybean meal. ‡Diet C = 0·14 wheat straw + 0·70 maize grain + 0·16 soybean meal.

nitrogen oxide emissions. Nitrate emissions from feed crops for diet C were 56% and 95% higher than those for diet MS and H, respectively.

Land occupation per kg of BWG was the highest for diet H (11·7 $m^2 \times yr$), which was 2·5 times higher that for diet MS (4·5 $m^2 \times yr$) and diet C (4·6 $m^2 \times yr$). For diet H, grassland contributed 0·71 to land occupation. For diet MS and diet C, all land occupation was cropland. As a result, the impacts expressed per ha of land occupied were 1·8–3·3 times lower for diet H than for diets MS and C (Table 5). Per ha of land occupied, acidification was the highest for diet MS, eutrophication and cumulative energy demand were the highest for diet C.

DISCUSSION

Differences among diets for environmental criteria

The three diets which have been studied are very different. Diet C is a feedlot diet that does not require forage production, compared to the other two diets. They also differ in BWG, which is the highest for diet C and the lowest for diet H. With these diets, the climate change impact per kg of BWG was 3·65, 4·56 and 4·74 kg CO_2 equiv. for diets C, H and MS, respectively, when taking into account C sequestration by soils (Doreau *et al*. 2011). Differences between diets are not ranked similarly for impacts studied in the current paper. The lowest energy demand/kg of BWG with diet MS was due to the lower energy demand for feed production. This is principally related to the higher DM yield for maize silage, as the whole plant is harvested. Although maize silage has a lower digestible energy/kg DM than maize grain, maize silage yielded higher digestible energy/ha than maize grain. Secondly, energy demand/ha of maize silage and maize grain produced in Brittany was lower than that of maize produced in other regions, as maize in Brittany is mainly fertilized with pig slurry. In the present study, the highest energy demand per kg of BWG for diet C was due to its high energy demand for crop-based feed production, although animals had the highest average BWG. The high energy demand per kg BWG for diet H was due to its low average BWG. These results confirm the important contribution of the crop production

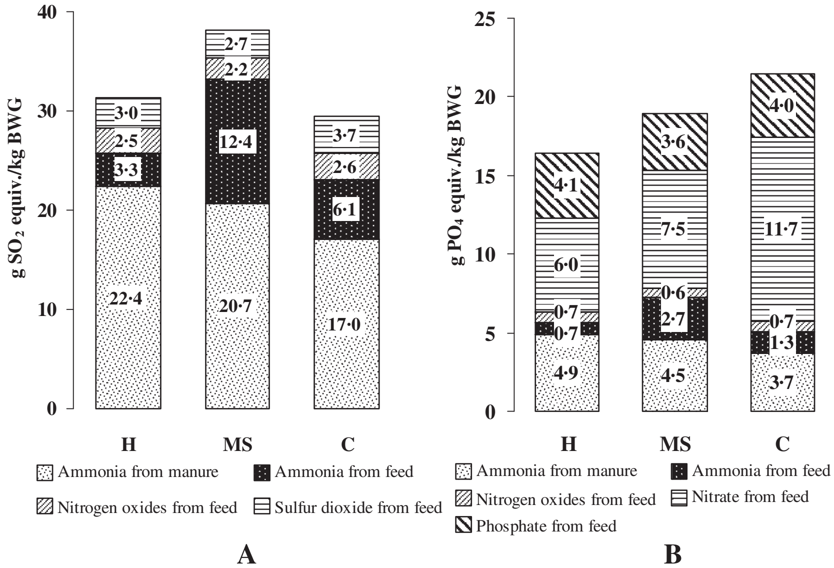

Fig. 2. Contribution of emitted substances from feed production and manure management to AP (A) and EP (B) arising from fattening systems based on hay (diet H*), maize silage (diet MS†) or concentrate (diet C‡), expressed per kg BWG. *Diet H = 0·49 natural grassland hay + 0·41 maize grain + 0·10 soybean meal. †Diet MS = 0·63 maize silage + 0·21 maize grain + 0·16 soybean meal. ‡Diet C = 0·14 wheat straw + 0·70 maize grain + 0·16 soybean meal.

Table 5. *Cumulative energy demand, AP, EP arising from fattening systems based on hay (H), maize silage (MS) or concentrate (C), expressed per ha of land occupied*

	Diet*		
	H	MS	C
Cumulative energy demand (GJ equiv.)	16·0	28·6	42·6
AP (kg SO_2 equiv.)	26·7	83·7	63·6
From feeds	7·6	38·2	26·8
From manure	19·1	45·6	36·8
EP (kg PO_4 equiv.)	14·1	41·8	46·4
From feeds	9·9	31·8	38·4
From manure	4·2	10·0	8·0

* Diet H = 0·49 natural grassland hay + 0·41 maize grain + 0·10 soybean meal; Diet MS = 0·63 maize silage + 0·21 maize grain + 0·16 soybean meal; Diet C = 0·14 wheat straw + 0·70 maize grain + 0·16 soybean meal.

stage to the total energy demand for cattle fattening, as shown by Ogino *et al.* (2004) and Pelletier *et al.* (2010). The contribution of feed transport to the farm was principally related to the transport of soybean meal, especially due to road transport in Brazil (Prudêncio da Silva *et al.* 2010) rather than to sea transport from Brazil to France, or even road transport in France.

In the current study, AP was mainly due to ammonia from manure (0·54–0·72) and EP was mainly due to feed production (0·70–0·83). This is in agreement with Ogino *et al.* (2004) for AP and with Pelletier *et al.* (2010) for EP. As discussed by Nguyen *et al.* (2010), it is not clear whether phosphate and nitrate emissions from feed production were taken into account for EP in Ogino *et al.* (2004). The highest AP/kg of BWG for diet MS can be explained by the use of pig slurry for maize grain and maize silage in Brittany, causing high ammonia emissions. The highest ammonia emission from manure management for diet H resulted from the lowest BWG of animals in this system. This was the opposite of diet C, for which AP was the lowest, even if contribution of ammonia from feed was higher for diet C than for diet H. The highest EP for diet C can be explained by the highest nitrate emission from feed, principally from maize grain production. In Aquitaine, maize grain production is a monoculture (AGRESTE 2006), i.e. maize is followed by maize, and as a result, there is a high risk of nitrate leaching (Basset-Mens *et al.* 2007). Diet H resulted in the lowest EP, even with the lowest BWG, because of the low ammonia and nitrate emissions from hay production.

Pelletier *et al.* (2010) found lower energy use and EP/kg BWG for a feedlot system rather than for a system where a period of grazing high-quality forage occurred before fattening. In the same way, Nguyen *et al.* (2010) found higher energy use, EP and AP/kg carcass weight for steers fed large amount of forages than for bulls fed a concentrate-based diet, but this effect is mainly due to the difference in fattening duration. The amount of fertilizers and other inputs may have differed between these studies; however, the major cause of the difference between these studies is the large difference in BWG between the feeding systems in Pelletier *et al*. (2010) and Nguyen *et al*. (2010), whereas the difference is lower in the current trial.

The three diets differed in the area of land used for production. A large area cultivated/kg of product is often considered detrimental, owing to the competition for land between grassland and crops dedicated to human food or biofuels. However, in Europe, hay-based diets for bull-fattening are used mainly in semi-mountain or mountain areas, and in this case grasslands may be in competition only with forest, which stores more carbon than grasslands but does not contribute to human food. More generally, pastures that occupy 0·26 of ice-free terrestrial surface of the planet often do not compete with crops for food and biofuels (FAO 2009), as soils or climate do not allow annual cropping.

Differences according to the method of calculation of impacts

Environmental impacts are calculated per kg of product (BWG, yielding meat in the present case), or per ha. Published data often consider only one of these expressions, depending on the priority: reducing the impact per kg of product or reducing the pollution for a limited territory. In the first case, the driver is meeting the present or future global demand. This allows the comparison of the efficiency of systems in a globalized world, and this is especially important when providing food for 9 billion people in 2050 is an objective. In the second case, the driver is reduction of pollution. Thus, there is a need for a land-based measurement. Most available data deal with the impact on climate change and show minor differences per kg of milk according to the production system (review by Martin *et al*. 2010). Few data have compared the two modes of expression for impacts other than climate change, especially for meat production; most comparisons deal with milk production. In four different countries (Sweden, Germany, the Netherlands and France), Cederberg & Mattson (2000), Haas *et al*. (2001), De Boer (2003) and van der Werf *et al*. (2009) showed that differences in EP and AP between conventional and organic dairy systems were small when impacts were expressed per kg milk, whereas impacts were much larger for conventional systems than for organic systems when they were expressed per ha. The same conclusion is drawn by Veysset *et al*. (2011) for energy consumption in conventional and organic beef systems, whereas there are few differences in energy consumption per ha among conventional beef systems (Veysset *et al*. 2010). The extent of differences between conventional and organic systems probably depends on differences in stocking rates and off-farm surfaces.

In the current study, when results are expressed per ha, diet H showed lower potential impacts on eutrophication, acidification and cumulative energy demand than the other two diets, due to its higher land occupation. A similar conclusion had previously been drawn for climate change (Doreau *et al*. 2011). It is thus unlikely that, with the H diet, local impacts as eutrophication reach levels that cause actual environmental damage, especially when this diet is practised in a semi-mountain area with a moderate livestock density.

CONCLUSIONS

The current analysis shows that the type of diet for fattening bulls strongly affects the bulls' environmental impacts. When impacts are expressed per kg of BWG, differences between diets may reach 0·25–0·35 according to the impact considered. It is not possible to define the 'best' diet, because the ranking between diets is not the same according to the impact category. Impacts can also be considered per surface unit. In this case the diet based on hay results in much lower impacts because it requires a larger area for the same production. The impact on climate change also depends whether carbon storage in soils is considered. It is concluded that stakeholders and policy makers need to integrate the different environmental issues, and that decisions that will be taken should depend on the priority, maintaining animal production or reducing the pollution on a given territory.

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Abstract

Life cycle assessment of cattle production: exploring practices and system changes to reduce environmental impacts

This thesis addresses the environmental impacts of cattle production systems. The first objective of this thesis was to analyse and compare the environmental impacts of suckler-beef and dairy production systems using attributional life cycle assessment (ALCA). Subsequently, the effects of mitigation practices for suckler-beef production systems were assessed. The second objective addressed methodology development by exploring possible consequences due to an increase in preference for grass-based milk using consequential LCA (CLCA).

For a suckler-beef production system, enteric methane fermentation was the main contributor to the climate change impact, and grassland production contributed most to other impacts (cumulative energy demand, eutrophication, acidification and land occupation). The suckler cow-calf herd substantially contributed to the impacts of the suckler-beef system. The most effective mitigation practice for the suckler-beef production system was decreasing calving age from 3 to 2 years. The use of lipids rich in omega-3 fatty acids in ruminant diets did not substantially affect the impacts of the suckler-beef production system. Simultaneous application of several compatible practices can substantially mitigate the impacts of the suckler-beef production system. The application of certain practices (e.g. reducing ungrazed grass losses, fattening heifers not used for replacement and reducing calving age) reduced land occupation. Alternative uses for the “released land”, e.g. the introduction of forest to sequester C into biomass, seems promising.

For dairy production systems, the assessment focused on a grass-based vs. maize-silage-based system, dual-purpose breed (Normande) vs. specialised breed (Holstein) and the effect of increasing milk yield per cow, using the ALCA approach. Independent of co-product handling methods, the impacts per kg of milk were lower with the maize-silage-based system and with Holstein cows (except for eutrophication). Increasing milk yield per cow by increasing feed energy intake and applying more intensive management (first calving at 2 years) decreased the impacts of milk and its beef co-product. The consequences of converting a maize-silage-based to a grass-based dairy farm in France to meet the increased domestic preference for grass-based milk were assessed using the CLCA approach. This farm conversion caused land-use change outside the dairy farm and thus substantially increased the impacts of the whole production system and the milk it produced.

Keywords: suckler-beef, dairy, production systems, attributional life cycle assessment (ALCA), consequential LCA (CLCA), land-use change, environmental impacts

Résumé

Analyse de cycle de vie de la production bovine : exploration de pratiques et de changements de système pour réduire les impacts environnementaux

Cette thèse porte sur l’étude des impacts environnementaux de systèmes de production de bovins. Le premier objectif était d’analyser et de comparer les impacts environnementaux de systèmes de production de viande et de lait par analyse de cycle de vie (ACV) attributionnelle. Les effets de pratiques d’atténuation de ces impacts ont été évalués pour les systèmes de production de viande. Le second objectif était un développement méthodologique afin d’explorer les conséquences possibles d’une préférence accrue pour un lait produit à base d’herbe, par ACV conséquentielle.

Dans un système de production de viande par le troupeau allaitant, le méthane entérique a été le principal contributeur à l’impact changement climatique, et la production de l’herbe a été la principale contributrice aux autres impacts (demande énergétique cumulée, eutrophisation, acidification, occupation du sol). L’atelier naisseur (vaches allaitantes et leurs veaux, génisses) a contribué de manière majeure aux impacts du système allaitant dans son ensemble. La pratique d’atténuation la plus efficace pour le système a été la diminution de l’âge au vêlage de 3 à 2 ans. L’utilisation de lipides riches en acides gras oméga-3 dans le régime a très peu affecté les impacts du système. L’application simultanée de plusieurs pratiques d’atténuation compatibles entre elles réduit sensiblement les impacts. L’application de pratiques telles que la réduction du gaspillage d’herbe, l’engraissement des génisses non utilisées pour le renouvellement et la diminution de l’âge au vêlage réduisent l’occupation du sol. Un usage alternatif des terres libérées tel que la plantation de forêt pour séquestrer du carbone dans la biomasse semble prometteur.

L’étude de systèmes de production de lait a été centrée sur les comparaisons de systèmes à base d’herbe ou d’ensilage de maïs, d’une race spécialisée (Holstein) ou mixte (Normande) et sur l’effet du niveau de production laitière par ACV attributionnelle. Quelle que soit la méthode d’attribution des impacts aux co-produits, les impacts par kg de lait ont été plus faibles pour les systèmes à base d’ensilage de maïs et pour les Holstein, sauf pour l’eutrophisation. L’accroissement de la production de lait par vache grâce à une consommation d’énergie accrue et au vêlage à 2 ans a permis de réduire les impacts du lait et de son co-produit viande. Les conséquences de la conversion d’une exploitation laitière utilisant beaucoup de maïs ensilage vers une exploitation utilisant de l’herbe comme unique source de fourrage pour répondre à une demande de lait produit à base d’herbe en France ont été évaluées par ACV conséquentielle. Cette conversion entraîne des changements notables de l’utilisation des sols en dehors de l’exploitation, et donc un fort accroissement des impacts du système dans son ensemble et du lait produit.

Mots-clés : troupeau allaitant, troupeau laitier, systèmes de production, analyse de cycle de vie (ACV) attributionnelle, ACV conséquentielle, changement d’usage des sols, impacts environnementaux