Animal (2013), **7:s2**, pp 212–219 © The Animal Consortium 2013
doi:10.1017/S1751731113000785

An LCA researcher's wish list – data and emission models needed to improve LCA studies of animal production

C. Cederberg[1,2†], M. Henriksson[3] and M. Berglund[1]

[1]SIK, the Swedish Institute for Food and Biotechnology, PO Box 5401, SE-402 29 Gothenburg, Sweden; [2]Department of Energy and Environment, Chalmers University of Technology, SE- 412 96 Gothenburg, Sweden; [3]Department of Rural Buildings and Animal Husbandry, Swedish University of Agricultural Sciences, PO Box 86, SE-23053 Alnarp, Sweden

(Received 3 February 2013; Accepted 28 March 2013)

The last decade has seen an increase in environmental systems analysis of livestock production, resulting in a significant number of studies with a holistic approach often based on life-cycle assessment (LCA) methodology. The growing public interest in global warming has added to this development; guidelines for carbon footprint (CF) accounting have been developed, including for greenhouse gas (GHG) accounting of animal products. Here we give an overview of methods for estimating GHG emissions, with emphasis on nitrous oxide, methane and carbon from land use change, presently used in LCA/CF studies of animal products. We discuss where methods and data availability for GHGs and nitrogen (N) compounds most urgently need to be improved in order to produce more accurate environmental assessments of livestock production. We conclude that the top priority is to improve models for N fluxes and emissions from soils and to implement soil carbon change models in LCA/CF studies of animal products. We also point at the need for more farm data and studies measuring emissions from soils, manure and livestock in developing countries.

Keywords: life-cycle assessment, carbon footprint, livestock production, greenhouse gas emissions, methodology

Implications

Life-cycle assessment (LCA) is a well-established method for analysing the environmental impact of agricultural production systems and in recent years it has been increasingly used for calculations of life cycle greenhouse gas emissions of meat, milk and eggs. There are large uncertainties associated with models used for emission estimates of nitrous oxide from soils and for carbon fluxes and emissions from land use and land use change. Improving emission estimates is absolutely essential to achieve better assessments of the many mitigation options suggested to reduce the livestock sector's impact on global warming.

Introduction

The rapidly growing global demand for meat and milk is garnering increased attention in the media and public discourse. The FAO report 'Livestock's Long Shadow' (Steinfeld *et al.*, 2006) highlighted the greenhouse gas (GHG) emissions associated with animal production, estimating these at close to 18% of global GHG emissions. About 38% of the global land area is used for agriculture; around two-thirds of this is used for livestock feed, mostly pastures but also feed crops on arable land (Foley *et al.*, 2011). Losses of nitrogen (N) and phosphorus (P) lead to severe eutrophication problems in many regions of the world; livestock production systems generally have lower nutrient efficiency than crop systems. Half the annual global N surplus (the input less the uptake in crops and grazing) is emitted as reactive N (ammonia and nitrate), while half is denitrified (Bouwman *et al.*, 2009). This implies that the animal sector is responsible for a major proportion of emitted reactive N. Several studies have highlighted the environmental impacts of the fast-growing global livestock sector (e.g. Steinfeld *et al.*, 2006; Pelletier and Tyedmers, 2010). Dietary shifts in high-income countries – a reduction of overall intake of animal products – are increasingly discussed as a necessary mitigation option, cf. Foley *et al.*, 2011; Garnett, 2011; Wirsenius *et al.*, 2011; Cederberg *et al.*, 2013.

Calculating and assessing the numerous environmental impacts from livestock production requires a holistic approach. Life-cycle assessment (LCA) is a well-established method for analysing the environmental impact caused by a product or a service. The basic principle is to follow a product through its entire life cycle, that is, to have a whole supply-chain perspective. The product system studied is delimited from the surrounding environment by a system boundary. The energy and material flows crossing the boundary are

† Present address: SIK, the Swedish Institute for Food and Biotechnology, PO Box 5401, SE-402 29 Gothenburg, Sweden. E-mail: christel.cederberg@sik.se

accounted for as input related (e.g. resources) and output related (e.g. emissions to water and air). LCA is now standardized according to ISO 14040 and 14044 (ISO 2006a and 2006b). The first LCAs were carried out in the 1970s from energy analyses. For example, Coca Cola investigated the consequences of switching from glass bottles to aluminium cans (Baumann and Tillman, 2004). In the 1990s, LCA started to be used in analysing agricultural systems, a European Union project to harmonize methodology was carried out (Audsley *et al.*, 1997).

In recent years, different GHG accounting methods have emerged. In the product-based perspective, these methods yield the product carbon footprint (CF), including the product's life-cycle GHG emissions. When calculating a CF, the principles of LCA are used, but the focus is on one major impact: the product's contribution to global warming. CFs of milk, meat and eggs are typically dominated by estimates of nitrous oxide and methane – with much greater uncertainty than estimates of carbon dioxide emissions from fossil fuels (Rypdal and Winiwater, 2001) – and emissions associated with land use.

Here we give an overview of methods for estimating GHG emissions presently used in LCA/CF studies of animal products. We discuss where methods and data availability for GHGs and N compounds most urgently need to be improved in order to produce more accurate environmental assessments of livestock production.

Emission profiles

GHG emissions from land use change (LUC) have been included in LCA/CF studies of agricultural products only in recent years because of lack of uniform methodology as well as emissions from land use (LU), for example, effects of different soil management methods. Setting aside GHG emissions from land use and land use change (LULUC), which we will discuss later, nitrous oxide (N_2O) and methane (CH_4) originating from biological processes in soils, manure and livestock dominate the emissions profile of animal products. Figure 1 shows a typical distribution of GHGs emitted from primary production (cradle to farm gate) of poultry meat, pork, beef and milk in the developed world, here exemplified by average EU-27 production according to Weiss and Leip (2012) not including land-related emissions.

Poultry production (chicken meat and eggs) is the only sector that has relatively low emissions of methane, whereas carbon dioxide (CO_2) from the use of fossil fuels represents a significant GHG flux as well as N_2O from soils. Feed production (including fertilizer use and production) and barn heating are important sources of these emissions. The CF of pork production is dominated by N_2O from manure application, loss of reactive N and synthetic N fertilizer production and use, and CO_2 from fossil fuels. Cattle production, resulting in milk and beef products, is dominated by CH_4 from enteric fermentation, typically representing around half of total emissions in primary production (Figure 1).

Findings on emissions from animal production in one system/climate cannot simply be generalized to others.

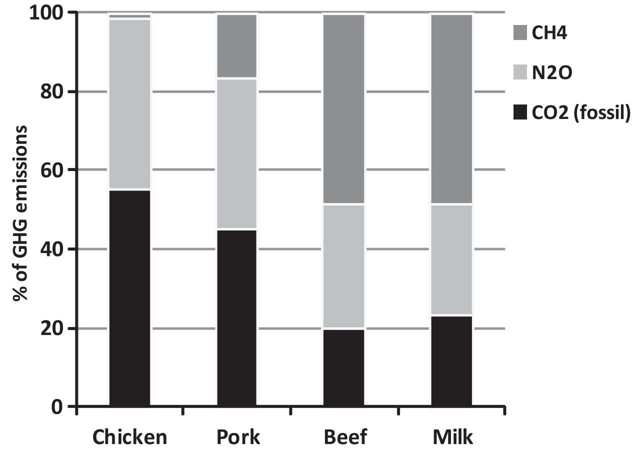

Figure 1 Typical distribution of greenhouse gases (%) in meat and milk production in developed regions (average EU-27) based on a study conducted by Weiss and Leip (2012). N.B. emissions from land use and land use change (LULUC) are not included.

In milk production in New Zealand, an outdoor pasture system as opposed to an indoor system with pronounced use of concentrate feed (which is typical for north European milk production), the CH_4 share is greater ($>$60%), while the N_2O share of the CF is similar to North European systems (Flysjö *et al.*, 2011). This is explained by a lower milk yield per cow in New Zealand, leading to higher emissions from enteric fermentation per kilogram milk but, on the other hand, lower emissions from fossil fuels because of the pasture-based feeding strategy. Thoma *et al.* (2012) estimated emissions from dairy farms in five US regions; their findings indicate that CH_4 from manure management represents a significantly higher share of the CF in the Midwest and Southwest because of the adoption of anaerobic lagoons and higher temperatures. Comparing a pasture-based beef system in South America with beef systems in western Europe reveals that CH_4 from enteric fermentation dominates the emission profile of South American production even more than demonstrated in Figure 1 – up to 75% of total emissions – whereas CO_2 from energy use stands for a minimal share of the CF of beef (Cederberg *et al.*, 2011).

Livestock and manure

Ruminant enteric fermentation

Enteric CH_4 is produced as a by-product of microbial fermentation in the rumen and large intestine of ruminants. When billons of rumen microbes break down cellulose into volatile fatty acids, hydrogen ions are released. To neutralize these ions, methanogenic bacteria form CH_4. The amount of CH_4 production is regulated by several parameters, for example, feed intake and structure, as well as nutrient composition. CH_4 production also leads to a loss of about 6.5% of the cattle's gross energy intake (IPCC, 2006). Numerous studies have been carried out to find dietary strategies to reduce CH_4 generation and energy loss while maintaining livestock productivity; the most promising alternatives appear to be adding fat to the diet, diets with higher starch content and use of some feed additives (Grainger and Beauchemin, 2011). Diets with a high share of forage imply more CH_4, but this also depends on forage digestibility (Patel *et al.*, 2011).

Several models predict enteric CH_4, from simple regression equations where CH_4 is linked to one variable (e.g. gross energy intake in IPCC Tier 2; IPCC, 2006) or several variables (e.g. Kirchgessner, 1995) to more complex mechanistic and stochiometric models that require significantly more detailed data (e.g feed data like digestion rate and content of soluble carbohydrates (Legesse *et al.*, 2011)). Simple regression equations are more user-friendly and included in many whole-farm models for calculating GHG emissions as well as used in LCA/CF studies.

Developing models that enable good predictions of enteric CH_4 emissions seems difficult. Although current detailed mechanistic and stochiometric models generally perform better than simpler regression equations, they should be used with care when creating and evaluating different mitigation strategies (Alemu *et al.*, 2011). Estimates of CH_4 vary from model to model, so the choice of model is fundamental to the final GHG estimate for milk and beef. According to Ellis *et al.* (2010), predictions by simple regression equations are generally poor, although they can be improved when some important dietary characteristics are included. Regression equations are mostly based on empirical studies of measured CH_4 emissions and can include older data from production systems, animal genetics and feed ingredients that are not currently used today, for example, dried forage instead of silage, cf. Jentsch *et al.* (2007).

The accuracy of emission models determines how well enteric CH_4 is calculated in LCA/CF studies of ruminant production. There is obviously a conflict in combining good predictive power with user-friendly models that are the first choice for most LCA researchers. Using LCA methodology to compare GHG emissions from different dietary or management scenarios requires models that can finely resolve the CH_4 differences among these. However, using mechanistic models requires detailed animal and feed data and the ability to run and evaluate a more complex model. Experts on ruminant diet and researchers working with environmental assessments of ruminant production systems need to collaborate closely to improve CH_4 analyses which otherwise risk of leading to invalid mitigation strategies.

Manure

Manure contributes not only to release of GHGs but also to eutrophying and acidifying compounds. Reliable data on N excretion rates are important when calculating manure emissions, since both N_2O and ammonia (NH_3) loss are modelled starting from N production in excreted faeces and urine. In LCA/CF studies, national excreted N values can be used, but preferably these input data should be calculated from the actual N in feed intake and in products that would improve emission estimates of NH_3 and N_2O from manure. Determining feed intake and thus N-excretion rates is a special challenge in ruminant production where grass silage with varying dry matter and N content is often the major feed ingredient. Further difficulties are associated with estimating intake from grazing animals. The EU project REDNEX is developing a model that can facilitate accurate predictions of quantities of N excreted by dairy herds with a standardized methodology in EU countries (www.rednex.fp7.eu). The model will separately predict faecal and urinary N excreted, have a database on N content of several feed ingredients, and be accessible on the Internet. This is good example of a tool that will simplify emission estimates from the manure of ruminants; such a tool would be useful for other livestock categories.

NH_3 emissions from manure are generally calculated with a mass flow approach following excreted N flow at different stages in the manure system: first divided between indoor maintenance/outdoor grazing, and, when excreted indoors, following the N flow further to storage and finally to manure application in the field (EMEP/EEA, 2010). NH_3 emission factors (EFs) are developed, mostly for European conditions, for different stages, manure types and application techniques. However, there are data gaps for some EFs, and for some world regions, there are no or very few background experiments supporting EFs for the specific manure systems and climate in that region. For example, very few data are available on ammonia loss from urine and dung in pastures under tropical conditions despite the fact that grazing is the major feeding system in beef production, which is rapidly growing in South America (Cederberg *et al.*, 2009). A few studies indicate high NH_3 loss from urine applied on bare soils and Brachiaria grasses in Brazil (Boddey *et al.*, 2004), considerably higher than what EFs of manure from pasture in temperate conditions indicate. Even in well-developed nations surprisingly little research on NH_3 emissions is published. Faulkner and Shaw (2008) conducted a review of $NH_{3,}$ EFs for animal agriculture in the United States, showing that many of the factors used were based on European literature that originated in studies of animal production systems with climate and techniques unlike those in the United States. The review found an urgent need for region-specific NH_3 EFs for US agriculture so that the impact of different NH_3 volatilization mitigation techniques can be accurately evaluated.

CH_4 emitted from slurry storage is a significant source of GHG from dairy and pork production. These emissions are typically calculated with Tier 2 models suggested by IPCC guidelines, including parameters for manure production (as volatile solids), a factor for the CH_4 producing capacity of different manure types, and a methane conversion factor (MCF) that is given as a default value for different manure systems and specific annual average temperatures in the range 10°C to 28°C. However, country-specific values are recommended to improve estimates. Recent investigations of CH_4 emissions from pig and cattle slurry storage in Mid Sweden (average annual temperature 8°C) showed MCF to be considerably lower, only 3% (Rodhe *et al.*, 2012), which implies a substantially lower manure storage emissions estimate than the IPCC suggested default value of 10% yields. Anaerobic digestion is considered to be one of the most promising mitigation techniques in agriculture, therefore it is important to have reasonably accurate parameter data for calculating GHG emissions from manure storage. If the emissions are not so high in colder climate zones because of lower methane formation in the manure, as indicated by this recent research, biogas production is not necessarily an effective measure for cutting livestock production's overall GHG emissions.

Nitrous oxide

Direct soil emissions

The main source of N_2O emitted from soils is denitrification. The biological processes in soils that form and release N_2O are complex, and emission levels depend on local conditions, such as climate and soil properties, but also on farming practices, for example, N-fertilizer rates and regimes. Soil N_2O loss is characterized by large spatial and temporal variations that are difficult to predict. A recent compilation of European field experiments in arable land and grassland showed emission levels ranging from 0.04 to 21.2 kg N_2O-N/ha and year (Rees *et al.*, 2012). N_2O soil emissions represent a large share of the total GHG emissions in feed crop cultivation (van der Middelaar *et al.*, 2013). N_2O is a very potent GHG, with a global warming potential (GWP) 300 times that of CO_2 over a 100-year timeframe, therefore even a few kilogram emitted in crop cultivation may lead to higher GHG emissions in the feed's life cycle than important production inputs such as fossil energy and synthetic N fertilizers. The great variability in N_2O emission levels and the uncertainty in emission estimates contribute to much of the uncertainty in GHG estimates of feed products.

Besides being a potent GHG, N_2O also contributes to ozone depletion. Emissions of other anthropogenic ozone-depleting substances, for example, chlorine- and bromine-containing halocarbons, are steadily declining and have dropped significantly since the 1980s, whereas the levels of N_2O emission are more stable (IPCC, 2007; Ravishankara *et al.*, 2009). Today, N_2O is the single most important substance among anthropogenic ozone-depletion emissions and will dominate in the future as ozone-depleting halocarbons are exhausted (Ravishankara *et al.*, 2009). N_2O is rarely included in assessments of the ozone-depletion potential in LCA, but the importance of considering N_2O has been recognized (Lane and Lant, 2012).

Simplified equations for calculating soil N_2O loss are generally used in LCA/CF studies, predominantly the IPCC Tier 1 model (IPCC, 2006), according to which N_2O emissions from managed soils are calculated from universal EFs multiplied by N input to soils via fertilizers, manure and crop residues, so-called direct N_2O emissions, and emissions of reactive N (ammonia and nitrate), so-called indirect N_2O emissions, respectively. These equations are simple to use, and the required input data are generally available. The disadvantage is that they fail to reflect any parameter other than N input, and they are not intended to be used at the field level to evaluate different management practices, for example, varying manure application techniques. In practice, the relation between N input and N_2O emissions may be weak. The LCA community acknowledges the weaknesses of this simplified approach, and efforts are made to better estimate soil N_2O emissions. For example, national adopted EFs have been used in LCA studies of feed and crop production (e.g. Biswas *et al.*, 2008; van der Middelaar *et al.*, 2013) and of milk products (e.g. Flysjö *et al.*, 2011).

There are several process-based models for simulating soil N_2O emissions, for example, the COUP model, DNDC (DeNitrification-DeComposition), DAYCENT (Daily Century Model) and CERES-EGC (Colorado State University, 2012; INRA, 2012; KTH, 2012; University of New Hampshire, 2012). CERES-EGC has been used to model N_2O emissions in LCAs of fava bean and cereals (Goglio *et al.*, 2012). The drawback of these advanced models is that they require detailed data on climate, soil and plant properties, and cultivation practices, and such detailed data are rarely attainable with reasonable effort in LCA studies.

LCA/CF studies of feed crop production would greatly benefit from better estimates of soil N_2O emissions, for example, via collaborations among professional users of advanced emission models and LCA modellers. The probable shortage of input data, for example, detailed data from weather stations and on soil properties, poses a challenge. In addition, site-specific conditions may be of less interest in LCA/CF studies as these often intend to assess characteristic cultivation systems in a region/district rather than crops from specific fields. Another alternative is to develop user-friendly and widely accepted models that include more aspects than merely N input, preferably parameters that are associated with agricultural management practices. Since the purpose of many LCA/CFs is to assess potential differences in environmental performance among management practices, cropping systems, and fertilization regimes, it would be advantageous if models were to predict the probable average emissions from the system studied rather than reproducing the emissions in a single year because of the particular weather conditions that year.

Indirect N_2O emissions

N_2O is also produced indirectly through volatilization of NH_3 and nitrate (NO_3^-) leaching. Consequently, good predictions of NH_3 loss in the whole manure-handling chain are of importance since as much as 20% to 30% of excreted N can be lost as NH_3. NO_3^- leaching is closely associated with feed crop cultivation, and the formation of N_2O from this leaching can contribute significantly to a feed product's CF. Van der Middelaar *et al.* (2013) evaluated methods and data sources for a feed's CF investigating six major feed ingredients with LCA data from three European countries. There was no consensus on models used for estimating NO_3^- leaching, which increased the uncertainty when comparing a feed's CF between the three countries. According to IPCC guidelines, a simplified method assumes that a fixed fraction of applied fertilizer and manure is lost. This is very far from the actual complex situation in which many factors determine the leaching. Here, the LCA researchers are in real need of improved methods.

LULUC

Soil carbon changes

Cumulatively since the pre-agricultural era, soils have lost 150 to 330 Gt CO_2 globally because of cultivation and disturbance (Smith, 2012). This is largely an effect of LUCs and poor management methods in agriculture. A meta-analysis indicates that soil C stocks decline (i.e. CO_2 is emitted) after LUC from pasture to forest plantation (−10%), native forest to forest plantation (−13%), native forest to crop (−42%) and pasture to crop (−59%). C stocks instead increase through soil C sequestration in conjunction with LUC when converting from native forest to pasture (+8%), crop to pasture (+19%), crop to forest plantation (+18%) and crop to secondary forest (+53%) (Guo and Gifford, 2002). A reasonable summary is that when ecosystems based on perennial vegetation such as pasture and forests are converted into annual crops, this leads to loss of soil organic carbon (SOC) and thus CO_2 emissions.

Effects of soil carbon changes (negative = CO_2 emissions; positive = soil C sequestration) because of LULUC are very rarely included in LCA/CF studies of agrifood products. Models for assessing soil organic matter dynamics are needed to provide reliable predictions of the size and change of soil C stocks for different soil types with varying management practices, for example, tillage practices, crop rotations, manure and fertilizer application and climate regimes. A number of models are available, but further development is needed in this dynamic research area. For example, soil structure should be incorporated in models, SOC content up to 1 m down should be modelled (most models only include top soils), and management factors, such as tillage, that influence crop residue and SOC depth distribution should be included (Stockmann *et al.*, 2013).

Vellinga and Hoving (2011) investigated the effect of dairy cattle feed by replacing pasture and grass silage with maize silage of high digestibility to reduce enteric CH_4 emissions. They modelled soil C changes when ploughing grassland for converting it into annual silage maize cropping to produce a feed with higher digestibility. However, when accounting for SOC changes because of decreasing soil C stocks when converting grassland into cropland and the loss of grassland's sequestration potential, CO_2 emissions were much larger than the annual mitigation from feeding more maize. LCA/CF studies of poultry and pork production have underestimated GHG emissions since mono-gastric production systems heavily depend on annual crops (grain and pulses) often grown in monoculture and annual cropping systems. However, ruminant production systems, where a large share of the feed base is pasture, have not been credited the C sequestration potential in permanent grassland; thus their emissions have been overestimated, at least relative to systems that rely more on annual crops. There is a now a strong interest in including SOC in LCA/CF studies. As is the case with estimates of N_2O from soils and CH_4 from enteric fermentation, it is very much a question of choosing a model that is reasonably user-friendly when it comes to the balance of data requirement and sufficient accuracy.

Deforestation

At present, global carbon emissions from LUC are estimated at 3.7 (±1.8) Gton CO_2 per year averaged for the decade 2002 to 2011, corresponding to 9% of total CO_2 emissions in 2011 (www.globalcarbon.com). These large amounts of GHGs have historically not been included in LCA/CF studies of animal products. In recent years, studies have been published that test and demonstrate different methods for attributing LUC-based emissions to meat and milk production systems, (e.g. Gerber *et al.*, 2010; Cederberg *et al.*, 2011; Ponsionen and Blonk, 2012; Weiss and Leip, 2012). Owing to the great complexity of the link between LU and deforestation, there is still no established consensus methodology on how to account for these impacts. Moreover, data gaps and uncertainties thwart robust estimates.

When a natural ecosystem, for example, a tropical rainforest, is transformed into new agricultural land, carbon is typically emitted in conjunction with forest burning, biomass decay and decreased soil carbon stocks. Forest carbon stock estimates have high variability. For example, investigations in Mato Grosso, Brazil, found a variability of more than two-fold in stock estimates (Morton *et al.*, 2011). A review of studies estimating indirect LUC (iLUCs) associated with biofuel production found a large variation in estimates of forest carbon stocks among studies, contributing to the uncertainty in iLUC accounting (DG; Energy, 2010). The size of the deforested area is another source of uncertainty. Brazil is considered to have one of the best monitoring programmes for deforestation, yet a 20% uncertainty in annual deforestation rate was found in the Mato Grosso study (Morton *et al.*, 2011). Increased knowledge of LU after deforestation is vital as carbon losses for cropland are potentially higher than for pasture, perennial plantation or secondary forests (Morton *et al.*, 2006). This is important not only when calculating emissions but also for improving our understanding of forces driving deforestation.

In addition to problems with the data, methodology issues need to be sorted out. If indeed we have a decent estimate of GHG emissions caused by a deforestation event, the emissions should be distributed over time and products. Allocation over time, over an 'amortisation period,' distributes the calculated emissions over, typically, 20 years of production. Beyond that, emissions from the deforestation event no longer burden the products from the deforested land. Allocation over products means distributing the emissions over the commercial products that are the result of the deforestation and produced on the converted land. Selected logging of valuable trees is often practised before the forest is burned. So far, only a few studies have considered timber outtake when distributing LUC emissions over products (Cederberg *et al.*, 2011; Ponsionen and Blonk, 2012), instead allocating all emissions to agricultural products (e.g. Gerber *et al*., 2010; Weiss and Leip, 2012).

Presently, various methods for including LUC emissions in LCA/CF studies of food products are presented in the scientific literature. Basically, there are now two quite opposite approaches: a product-based and a land-based approach. The product-based approach puts an LUC factor on product(s) that are directly responsible for deforestation based on area expansion over a given time period, (cf. Gerber *et al*., 2010; Cederberg *et al*., 2011; Weiss and Leip, 2012) which entails that soybeans and beef produced in South America have quite high LUC emissions (since soybeans and pasture for beef have led to expanding agricultural land) while European produced cereal crops have no LUC emissions (as cropland does not expand in Europe). The argument for a land-based approach is that any occupation of an area of land is responsible for deforestation, no matter if it takes place in Brazil or Europe, and therefore a general LUC factor should be put on land in agricultural production (Schmidt *et al*., 2012).

Applying these two approaches can give very different results in LCA/CF studies as exemplified by Flysjö *et al.* (2012) comparing GHG emissions from conventional and organic milk including LUC emissions. Organic milk showed a lower CF (compared with a high-yielding conventional system) when LUC-factors for soymeal were included, as less soy per kilogram milk was used in the organic system. However, when assuming that all land occupation is responsible for LUC (land-based approach), the organic system had a higher CF (compared with the high-yielding conventional system), as more land is required to produce organic milk. Owing to the lack of a consensus methodology and research work in progress in testing methods, it is important that emissions associated with LUC are presented separately from the other GHGs and that underlying assumptions and methods are explained transparently.

Fossil fuels

Calculating GHG emissions from fossil fuels, most importantly diesel fuel and synthetic fertilizers, is a relatively simple operation as there are reliable and relatively certain data on emissions associated with production and use of these inputs. However, difficulties arise in how to get accurate input data on the quantities used in different animal production systems, where feed production, processing and transports are highly dependent on fossil fuels. When GHG emissions are reported in national inventories, use of energy in the agriculture sector is listed in the energy sector, making it difficult to separate the agricultural sector's use and emissions with high certainty. In most LCA/CF studies, farm energy use is modeled and a critical review of the accuracy of input data is important. In a Danish study, which measured actual diesel fuel use on commercial farms to validate a farm energy model, Dalgaard *et al.* (2001) found a systematic underestimate of energy used for transportation and handling of roughage crops and manure in present models of livestock farms.

Production of synthetic N fertilizer represents a major energy cost and GHG emission source in feed crop cultivation. The data on process-based emissions are relatively reliable compared with the state of knowledge of N fertilizer rates at commercial farms. Henriksson *et al.* (2011) analysed variations in production parameters in Swedish milk production using data from more than 900 dairy farms, showing a variation in N fertilizer rates between 0 and 250 kg N/ha. Higher livestock density means greater access to manure, and therefore lower need for N fertilizers, but they found no correlation between N fertilizer rate and livestock density.

Trade of animal products, as well as cereals and oilseed for feed, has increased significantly over the last two decades. This development is projected to continue (Guyomard *et al*., 2013). Increased trade implies increased transports and processing of feedstuff; there is an obvious need for research on the effect on overall energy use in animal production, most importantly for pork and poultry.

Conclusions

The last decade has seen an increase in environmental systems analysis of livestock production, resulting in a significant number of studies with a holistic approach often based on LCAs. The growing public interest in global warming has added to this development; guidelines for CF accounting have been developed, including for GHG accounting of animal products. The growing literature on the environmental impact of livestock production has contributed significantly to improved knowledge of emissions and resource use in different production systems, environmental hot-spots in the animal food chain, and where improvements are most urgent.

We are entering a new phase in the use of environmental systems analysis in agricultural production. The past 10 years have given us a general broad understanding of the 'big picture' of GHG emissions from the livestock sector. We now need to deepen our understanding of biological processes in soil, manure and livestock and to implement this knowledge in models that more accurately describe processes and emissions. Improving our emission estimates is absolutely essential to achieve better assessments of the many mitigation options suggested in the livestock sector. Policy makers need such assessments to develop appropriate policies, for example, when choosing technologies to support or designing tax policy for GHG emissions from the agriculture sector. Consumers and business-to-business communications increasingly rely on CF information.

We believe that the top priority is to improve models for N fluxes and emissions from soils and to implement soil carbon change models in LCA/CF studies of animal products. The rough models generally used to estimate soil N_2O (mostly the IPCC guidelines Tier 1) fail to accurately distinguish the mitigation potentials of N management alternatives in crop production systems. In addition, loss of reactive N in the form of ammonia and nitrate is extensive from many animal production systems, causing environmental impacts that are far-reaching. Implementing adequate measures to decrease these impacts requires increasing our understanding of N emissions from soils and manure. Enteric CH_4 is a major

GHG from global animal agriculture, however we believe that emissions models used are less uncertain and better with respect to CH_4 than the ones estimating reactive N. Estimates of CO_2 from LULUC is obviously a very dynamic research field, fueled by interest in assessing biofuel production, and also of great interest for LCA/CF studies of animal products.

Industrialized nations are over-represented in studies of GHG emissions from agriculture, biasing global estimates. Climate, soils, management techniques and other factors that affect emissions differ significantly among regions. There is a great need for more and better data on fundamental farm management (e.g. manure handling methods, feed rations) in developing countries as well as more studies measuring emissions from soils, manure and livestock beyond industrialized nations. Future growth in agricultural production is predicted to take place in the developing world, and better basic knowledge of farm management methods and nutrients and GHG emissions is vital here.

Even if we manage to improve GHG and N emission models in agriculture, we will never fully succeed to include the overall complexity that is inherent in biological systems in soils and livestock. Further uncertainties and variation in production data from livestock systems cannot be fully avoided since agricultural production often takes place on many small production sites (farms) as opposed to industrial production. We need to use more statistical analysis in LCA/CF studies of agricultural production to handle the uncertainty and variations, and those who interpret and use the results must better understand that LCA/CF studies cannot deliver 'exact' numbers on environmental impacts; estimate ranges need to be interpreted and understood when results are presented.

We are convinced that more cooperation among LCA/CF modellers and specialist agriculture/livestock researchers is essential to further improve the holistic understanding of the environmental impact of agriculture. We need to recognize that LCA/CF methodology, software and databases have developed extensively over the past year.

Finally, despite the large uncertainties associated with estimates of biogenic GHG emissions (N_2O, CH_4 and non-fossil CO_2), there is robust, consistent information from the many LCA/CF studies published so far on how to reduce the environmental impact from livestock production: 1. Reduce and stop deforestation associated with expansion of agricultural land for feed crops and pasture as this is associated with large GHG emissions and also considerable biodiversity loss; 2. Improve N management to significantly reduce reactive N emissions and input of synthetic fertilizers that result in GHG emissions and also eutrophication problems; and 3. Reduce the use and dependency of fossil fuels in feed cultivation, processing and transports to reduce CO_2 emissions and the extraction of fossil resources.

Acknowledgement

We want to acknowledge the help with language editing from Paulina Essunger.

This paper was published as part of a supplement to *animal*, publication of which was supported by the Greenhouse Gases & Animal Agriculture Conference 2013. The papers included in this supplement were invited by the Guest Editors and have undergone the standard journal formal review process. They may be cited. The Guest Editors appointed to this supplement are R. J. Dewhurst, D. R. Chadwick, E. Charmley, N. M. Holden, D. A. Kenny, G. Lanigan, D. Moran, C. J. Newbold, P. O'Kiely, and T. Yan. The Guest Editors declare no conflict of interest.

References

Alemu AW, Dijkstra J, Bannink A, France J and Kebreab E 2011. Rumen stoichiometric models and their contribution and challenges in predicting enteric methane production. Animal Feed Science and Technology 166–167, 761–778.

Audsley E, Alber S, Clift R, Cowell S, Crettaz P, Gaillard G, Hausheer J, Jolliet O, Kleijn R, Mortensen B, Pearce D, Roger E, Teuleon H, Weidema B and van Zejts H 1997. Harmonisation of environmental life cycle assessment for agriculture. Final Report of Concerted Action AIR3-CT94-2028, Silsoe Research Institute, Bedford, UK.

Baumann H and Tillman AM 2004. The Hitch Hiker's guide to LCA – an orientation in life cycle assessment methodology and application. Studentlitteratur, Lund, Sweden.

Biswas WK, Barton L and Carter D 2008. Global warming potential of wheat production in Western Australia: a LCA. Water and Environment Journal 22, 206–216.

Boddey RM, Macedo R, Tarré RM, Ferreira E, de Oliveira OC, Rezende CP, Cantarutti RB, Pereira JM, Alves BJR and Urquiaga S 2004. Nitrogen cycling in the *Brachiaria* pastures: the key to understanding the process of pasture decline. Agriculture, Ecosystems and Environment 103, 389–403.

Bouwman AF, Beusen AHW and Billen G 2009. Human alteration of the global nitrogen and phosphorus soil balances for period 1970–2050. Global Biogeochemical Cycles 23, 1–16.

Cederberg C, Meyer C and Flysjö A 2009. Life cycle inventory of greenhouse gas emissions and use of land and energy in Brazilian beef production. SIK-report No 792, Swedish Institute for Food and Biotechnology, Gothenburg, Sweden.

Cederberg C, Hedenus F, Wirsenius S and Sonesson U 2013. Trends in greenhouse gas emissions from consumption and production of animal products in Sweden – implications for a long-term climate target. Animal 7, 330–340.

Cederberg C, Persson M, Neovius K, Molander S and Clift R 2011. Including carbon emissions from deforestation in the carbon footprint of Brazilian beef. Environmental Science & Technology 45, 1773–1779.

Colorado state university 2012. DayCent: Daily Century Model. Retrieved on January 10, 2013, from www.nrel.colostate.edu/projects/daycent/.

Dalgaard T, Halberg N and Porter JR 2001. A model for energy use in Danish agriculture used to compare organic and conventional farming. Agriculture, Ecosystems and Environment 87, 51–65.

DG Energy 2010. The impact of land use change on greenhouse gas emissions from biofuels and bioliquids. Literature Review. An In-house Review Conducted for DG Energy as Part of the European Commission's Analytical Work on Indirect Land Use Change.

Ellis JL, Bannink A, France J, Kebreab E and Dijkstra J 2010. Evaluation of enteric methane prediction equations for dairy cows used in whole farm models. Global Change Biology 16, 3246–3256.

EMEP/EEA 2010. Emission Inventory Guidebook 2009. Retrieved September 30, 2012, from http://eea.europa.eu/emep-eea-guidebook

Faulkner WB and Shaw BW 2008. Review of ammonia emission factors for US animal agriculture. Atmospheric Environment 42, 6567–6574.

Flysjö A, Cederberg C, Henriksson M and Ledgard S 2012. Interaction between milk and beef production and emissions from land use change – critical considerations in life cycle assessment and carbon footprint studies of milk. Journal of Cleaner Production 28, 132–142.

Flysjö A, Henriksson M, Cederberg C, Ledgard S and Englund J-E 2011. The impact on various parameters on the carbon footprint of milk in New Zealand and Sweden. Agricultural Systems 104, 459–469.

Foley JA, Ramankutty N, Brauman KA, Cassidy ES, Gerber JS, Johnston M, Mueller ND, O'Connell C, Ray DK, West PC, Balzer C, Bennett EM, Carpenter SR, Hill J, Monfreda C, Polasky S, Rockström J, Sheehan J, Siebert S, Tilman D and Zaks D P M 2011. Solutions for a cultivated planet. Nature 478, 339–342.

Garnett T 2011. What are the best opportunities for reducing greenhouse gas emissions in the global food system? Food Policy 36, S23–S32.

Goglio P, Colnenne-David C, Di Bene C, Bosco S, Laville P, Roche R, Ragaglini G, Doré T, Mazzoncini M, Gabrielle B and Bonari E. 2012. Soil, climate and cropping system effects on N_2O accounting in the LCA of faba bean and cereals. Proceedings of the 8th International Conference on Life Cycle Assessment in the Agri-food Sector, Saint-Malo, France, pp. 155–160.

Gerber P, Vellinga TV, Opio C, Henderson B and Steinfeld H 2010. Greenhouse gas emissions from the dairy sector – a life cycle assessment. Food and Agriculture Organization of the United Nations, Rome.

Grainger C and Beauchemin KA 2011. Can enteric methane emissions from ruminants be lowered without lowering their production? Animal Feed Science and Technology 166–167, 308–320.

Guo LB and Gifford M 2002. Soil carbon stocks and land use change: a meta analysis. Global Change Biology 8, 345–360.

Guyomard H, Manceron S and Peyrand J-L 2013. Trade in feed grains, animals and animal products: current trends, future prospects and main issues. Animal Frontier 3, 14–18.

Henriksson M, Flysjö A, Cederberg C and Swensson C 2011. Variation in carbon footprint of milk due to management differences between Swedish dairy farms. Animal 5, 1474–1484.

INRA 2012. Agro-ecosystem Model CERES-EGC. Retrieved January 10, 2013, from http://www4.versailles-grignon.inra.fr/egc_eng/Productions/Softwares-Models/CERES-EGC

IPCC 2006. Emissions from livestock and manure management (chapter 10). In Guidelines for national greenhouse gas inventories – volume 4 agriculture, forestry and other land use (ed. HS Eggleston, L Buendia, K Miwa, T Ngara and K Tanabe), 87pp. National Greenhouse Gas Inventories Program IGES, Japan.

IPCC 2007. Climate change 2007: the physical science basis. In Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change (ed. S Solomon, D Qin, M Manning, Z Chen, M Marquis, KB Averyt, M Tignor and HL Miller), 996pp. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

ISO 2006a. Environmental management – life cycle assessment – principles and framework. ISO 14040:2006(E). International Organization for Standardization, Geneva, Switzerland.

ISO 2006b. Environmental management – life cycle assessment – requirements and guidelines. ISO 14044:2006(E). International Organization for Standardization, Geneva, Switzerland.

Jentsch W, Schweigel M, Weissbach F, Scholze H, Pitroff W and Derno M 2007. Methane production in cattle calculated by the nutrient composition of the diet. Archives of Animal Nutrition 61, 10–19.

Kirchgessner M, Windish W and Müller HL 1995. Nutritional factors for the quantification of methane production. In Proceedings of the Eighth International Symposium on Ruminant Physiology.

KTH 2012. Coup Model – Coupled Heat And Mass Transfer Model for Soil-Plant-Atmosphere System. Retrieved January 10, 2013, from http://www2.lwr.kth.se/Vara%20Datorprogram/CoupModel/

Lane J and Lant P 2012. Including N_2O in ozone depletion models for LCA. International Journal of Life Cycle Assessment 17, 252–257.

Legesse G, Small JA, Scott SL, Crow GH, Block HC, Alemu AW, Robins CD and Kebreab E 2011. Predictions of enteric methane emissions for various summer pasture and winter feeding strategies for cow calf production. Animal Feed Science and Technology 166–167, 678–687.

Morton DC, Sales MH, Souza CM and Griscom B 2011. Historic emissions from deforestation and forest degradation in Mato Grosso, Brazil: 1) source data uncertainties. Carbon Balance and Management 6, 1–13.

Morton DS, DeFries R, Shimabukuro YE, Anderson LO, Arai E, del Bon Espirito-Santo F, Freitas R and Morisette J 2006. Cropland expansion changes deforestation dynamics in the southern Brazilian Amazon. PNAS 103, 14637–14641.

Patel M, Wredle E, Börjesson G, Danielsson R, Iwaasa AD, Spörndly E and Bertilsson J 2011. Enteric methane emissions from dairy cows fed different proportions of highly digestible grass silage. Acta Agriculturae Scandinavica, Section Animal Science 61, 128–136.

Pelletier N and Tyedmers P 2010. Forecasting potential environmental costs of livestock production 2000-2050. PNAS 107, 18371–18374.

Ravishankara AR, Daniel JS and Portmann RW 2009. Nitrous oxide (N_2O): the dominant ozone-depleting substance emitted in the 21st century. Science 326, 123–125.

Rees RM, Augustin J, Alberti G, Ball BC, Boeckx P, Cantarel A, Castaldi S, Chirinda N, Chojnicki B, Giebels M, Gordon H, Grosz B, Horvath L, Juszczak R, Klemedtsson ÅK, Klemedtsson L, Medinets S, Machon A, Mapanda F, Nyamangara J, Olesen J, Reay D, Sanchez L, Sanz Cobena A, Smith KA, Sowerby A, Sommer M, Soussana JF, Stenberg M, Topp CFE, van Cleemput O, Vallejo A, Watson CA and Wuta M 2012. Nitrous oxide emissions from European agriculture; an analysis of variability and drivers of emissions from field experiments. Biogeosciences 1, 9259–9288.

Rodhe L, Abubaker J, Ascue J, Pell M and Nordberg Å 2012. Greenhouse gas emissions from pig slurry during storage and after field application in northern European conditions. Biosystems Engineering 113, 379–394.

Rypdal K and Winiwarter W 2001. Uncertainties in greenhouse gas emission inventories – evaluation, comparability and implications. Environmental Science Policy 4, 107–116.

Schmidt JH, Reinhard J and Weidema BP 2012. A model for indirect land use change. Proceedings from 8th International Conference on Life Cycle Assessment in the Agri-Food Sector. Saint-Malo, France, pp. 245–251.

Smith P 2012. Soils and climate change. Current Opinion in Environmental Sustainability 4, 539–544.

Steinfeld H, Gerber P, Wassenaar T, Castel V, Rosales M and de Haan C 2006. Livestock's long shadow – environmental issues and options. Food and Agriculture Organization of the United Nations, Rome.

Stockmann U, Adams MA, Crawford JW, Field DJ, Henakaarchchi N, Jenkins M, Budiman Minasnya, McBratney AB, de Remy de Courcelles V, Singh K, Wheelera I, Abbott L, Angers DA, Baldock J, Birde M, Brookes PC, Chenu C, Jastrow JD, Lal R, Lehmann J, O'Donnell AG, Parton WJ, Whitehead D and Zimmermann M 2013. The known, known unknowns and unknowns of sequestration of soil organic carbon. Agriculture, Ecosystem and Environment 164, 80–99.

Thoma G, Popp J, Shonnard D, Nutter D, Matlock M, Ulrich R, Kellogg W, Kim DS, Neiderman Z, Kemper N, Adom F and East C 2012. Regional analysis of greenhouse gas emissions from USA dairy farms: a cradle to farm-gate assessment of the American dairy industry circa 2008. International Dairy Journal, http://dx.doi.org/10.1016/j.idairyj.2012.09.010

University of New Hampshire 2012. The DNDC Model. Retrieved January 10, 2013, from http://www.dndc.sr.unh.edu/

Van der Middelaar CE, Cederberg C, Vellinga TV, Van der Werff HMG and Boer IJM 2013. Exploring variability in methods and data sensitivity in carbon footprints of feed ingredients. International Journal of Life Cycle Assessment 18, 768–782.

Vellinga TV and Hoving IE 2011. Maize silage for dairy cows: mitigation of methane emissions can be offset by land use change. Nutrient Cycling in Agroecosystems 89, 413–426.

Weiss F and Leip A 2012. Greenhouse gas emissions from the EU livestock sectors: a life cycle assessment carried out with the CAPRI model. Agriculture, Ecosystems and Environment 149, 124–134.

Wirsenius S, Hedenus F and Mohlin K 2011. Greenhouse gas taxes on animal food products: rationale, tax scheme and climate mitigation effects. Climatic change 108, 159–184.