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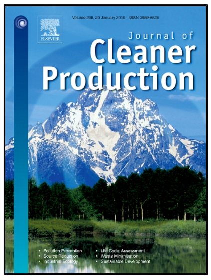

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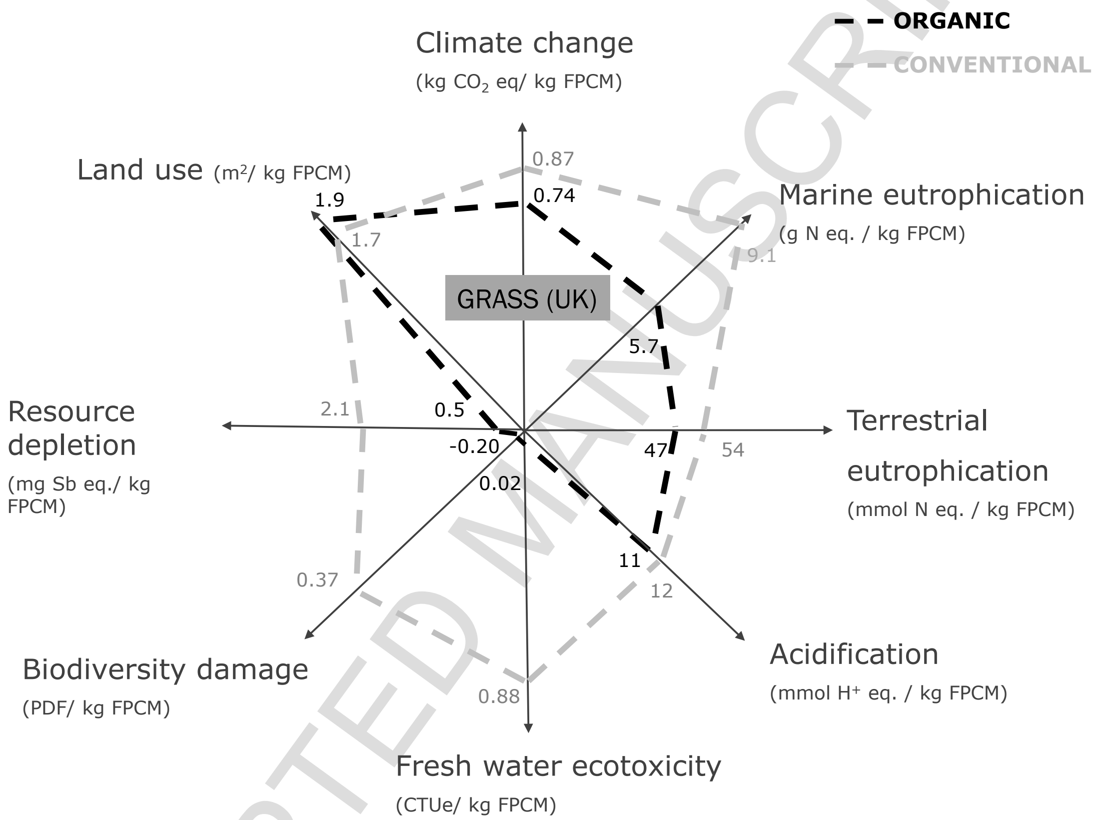
Graphical abstract



The importance of including soil carbon changes, ecotoxicity and biodiversity impacts in environmental life cycle assessments of organic and conventional milk in Western Europe

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Abstract

Estimates of soil carbon changes, biodiversity and ecotoxicity have often been missing from life cycle assessment based studies of organic dairy products, despite evidence that the impacts of organic and conventional management may differ greatly within these areas. The aim of the present work was therefore to investigate the magnitude of including these impact categories within a comprehensive environmental impact assessment of organic and conventional dairy systems differing in basic production conditions. Three basic systems representative of a range of European approaches to dairy production were selected for the analysis, i.e. (i) low-land mixed crop-livestock systems, (ii) lowland grassland-based systems, (iii) and mountainous systems. As in previous publications, this study showed that when assessing climate change, eutrophication and acidification impact organic milk has similar or slightly lower impact than conventional, although land-use is higher under organic management. Including soil carbon changes reduced the global warming potential by 5-18%, mostly in organic systems with a high share of grass in the ration. The impacts of organic milk production on freshwater ecotoxicity, biodiversity and resource depletion were 2, 33 and 20% of the impacts of conventional management, respectively, across the basic systems considered. The study highlights the importance of including biodiversity, ecotoxicity and soil carbon changes in life cycle assessments when comparing organic and conventional agricultural products. Furthermore, the study shows that including more grass in the ration of dairy cows increases soil carbon sequestration and decreases the negative impact on biodiversity.

Keywords: organic; biodiversity; dairy; ecotoxicity; LCA; soil carbon

1. Introduction

Meat and dairy products are responsible for a large proportion of the environmental impacts arising from food consumption (Ritchie et al., 2018). At the European level, climate policy stipulates that agriculture should contribute to climate change mitigation (EC, 2016) through protection and enhancement of soil organic carbon among other issues (EC, 2011). Likewise, the EU Biodiversity strategy 2020 (EC, 2011) has called for initiatives to halt the loss of biodiversity. Research has shown that organic farming can make a positive contribution in such areas (Tuck et al., 2014; Gattinger et al., 2012; Fliessbach et al., 2010; Hole et al., 2005; Bengtsson et al., 2005) although many product-level comparisons of environmental impact exclude any assessment of changes in on-farm biodiversity or soil carbon (Meier et al., 2015).

The lack of reported impacts concerning soil carbon changes, biodiversity, and ecotoxicity aspects is reflected in recent LCA studies of organic and non-organic milk production. (Salou et al., 2017; Hietala et al., 2015; Guerci et al., 2013; van der Werf et al., 2009 amongst others). Of all these studies only two include ecotoxicity (van der Werf et al., 2009; Salou et al., 2017), two covered biodiversity (Guerci et al., 2013; Müller-Lindenlauf et al., 2010), and two covered soil carbon changes (Kristensen et al, 2011; Guerci et al., 2013). None was found that simultaneously included ecotoxicity, biodiversity and soil carbon changes. One included both soil carbon changes and biodiversity, but focusing only on conventional milk (Battini et al., 2016). Although these aspects are recognized as being important, and may have particular relevance for organic farming, the EC Product Environmental Footprint Initiative (PEF) (EC, 2013) highlights that current Life Cycle Assessment (LCA) methodologies are not sufficiently developed for such areas to be confidently included in an assessment. Thus, it is decided that the impact of soil carbon sequestration in managed soils should be excluded when assessing GHG emissions until further agreed methods are available (EC, 2018a), and that biodiversity impacts, if relevant, should be reported as supplementary information outside the LCA impact categories (EC, 2018a). In the PEF for dairy products (EC, 2018b), examples are given of reporting impacts on biodiversity that might be used, e.g. participation in biodiversity schemes or share of semi-natural habitats of total farmland, all methods that are difficult to operationalize in a product-oriented assessment.

Petersen et al. (2013) suggested a methodology to include soil carbon changes in the climate change category, which is implemented in e.g. Knudsen et al. (2014) and Mogensen et al. (2014). In addition, Knudsen et al. (2017) proposed a method to include biodiversity impacts in assessment, based on plant species diversity as a proxy, which has been implemented by Parajuli et al. (2017). The characterization factors in Knudsen et al. (2017) was based on a unique dataset derived from field recording of plant species diversity in farmland across six European countries. Furthermore, Peña et al. (under review) have recently developed ecotoxicity characterization factors related to agricultural production. Thus, LCA based methods are available taken these aspects into account, but the magnitude of these impacts in different systems are not reported and it is not clear under which basic production conditions it is important to include these impacts in assessing possible differences between organic and conventional milk.

The main aim of this study was to investigate under which conditions the inclusion of a broader range of impacts such as soil carbon, biodiversity and ecotoxicity is important, when assessing the environmental impact of organic and conventional milk under different production conditions in Western Europe.

2. Materials and methods

2.1 The three selected European dairy systems

Three basic systems was selected for this analysis: mixed crop-livestock lowland systems (as found in Denmark), lowland grassland-based systems (as found in UK), and mountainous systems (in Austria). Each system is described in more detail in the following sections.

2.1.1 The mixed dairy system in Denmark

The mixed dairy system was represented by typical Danish production conditions for conventional and organic dairy systems, respectively. An overview of the mixed dairy systems is given in Tables 1 and 2. Data on herd size, farm size and milk yield were gathered from Statistics Denmark (Statistics Denmark, 2015). The mortality rate for young stock was estimated according to Danish statistics (Landbrugsinfo, 2015). Land use and crop yield were from Kristensen (2015), and application of manure and fertilizer were from the Danish regulation regarding fertilization and harmony rules (NaturErhvervsstyrelsen, 2015). Pesticides and doses of active ingredients (a.i.) used for the different crops were based on Ørum and Samsøe-Petersen (2014). The manure management system was assumed 100% slurry for cows and 75% slurry and 25% deep litter for replacement animals. The slurry was stored in tanks with cover, and there was no grazing for dairy cows in the conventional system (Mogensen et al., 2015).

2.1.2 The grassland-based system in UK

The grassland-based systems were represented by typical UK grassland-based dairy systems, for conventional and organic management respectively (see Tables 1 and 2). Herd size, farm size, milk yield, land use and crops yields were based on data from a range of sources (Moakes et al., 2015; Lampkin et al., 2012; The Professional Nutrient Management Group, 2015). Culling rate, days at pasture, and fertilization of forage area data were based on a recent study by the Agriculture and Horticulture Development Board (AHDB Dairy, 2014). Mortality rates were assumed similar to Danish systems. The yield of forage areas were based on industry guidelines (The Professional Nutrient Management Group, 2015) and expert evaluations (L.G. Smith). The manure management system was assumed 100% slurry for cows and 75% slurry and 25% deep litter for replacement animals. While slurry is often stored in tanks without a cover in the UK, for the purpose of this study it was assumed that slurry was kept in tanks with a cover, since it is assumed that this will be more common in future. The strategy for pesticide application per crop was assumed similar to those in the Danish system.

2.1.3. The mountainous system in Austria

The characteristics of the mountainous systems were represented by typical Austrian dairy systems (see Tables 1 and 2). Data were from IACS (2017) statistical data and expert evaluations. The replacement rates are similar to those in grassland-based systems. With regard to manure management systems, it was assumed that 40% was slurry and 60% was solid manure for both categories of animals (cows and replacement heifers). The storage of slurry was assumed to be in tanks with covers. The strategies for pesticide application per crop were assumed similar to those used in the mixed system.

2.2. Data and data quality

As mentioned, the basic data that describe the different systems were derived from a number of sources. The farm types investigated are therefore constructed for the purpose of this study and do not represent data collected from actual farming systems. The most important uncertainty is assumed the amount of input in the systems and the obtained forage yield per ha. However, in order to ensure coherence between input, output and amount of home grown feed - which is strongly related to the environmental impact - the same biological models were used across countries as described in section 2.3.2. Likewise, on-farm emissions were modelled using the same models across countries.

2.3. Life Cycle Assessment approach

The Life Cycle Assessment (LCA) approach considered impacts up-to the farm gate (i.e. the distribution, consumption and disposal/recycling of products was not considered). The ILCD 2011 Midpoint+ V1.06 was used for the impact assessment by means of the software tool SimaPro version 8.3 by PRé Consultants and the Ecoinvent 3.3 database.

2.3.1 Goal and scope

Functional unit and system boundaries

The functional unit in this work was 1 kg of milk (fat and protein corrected with 4% fat and 3.3% protein) at the farm gate. Generally, most of the environmental impact related to dairy products occurs before the milk leaves the farm (Thoma et al., 2013), so with this functional unit the major differences related to different production systems are captured. The life cycle of each dairy system was broken down into five components: 'Enteric fermentation', 'Electricity at the dairy farm', 'Home produced feed and manure management', 'Transport', 'Bought in feed'. The foreground processes included in the analysis were home-grown feed and milk production. The background processes included accounted for the production and transport of inputs, i.e. concentrate, fertilizers, pesticides, diesel, fuels, seeds and electricity. The energy and material inputs are traced back to the extraction of resources. Emissions and manure production from each life cycle stage are quantified. Capital equipment such as manure tank, roads, farm tractors, construction of stables, milk cooling systems, as well as the medicines used in stable were excluded from the analysis because they typically contribute less than 1% to the total environmental impacts of milk production systems (Dalgaard et al., 2014).

Impact categories

Guidelines for reporting the product environmental footprint (PEF) (EC, 2015) of dairy products were applied in the selection of impact categories. Seven out of eight impact categories recommended for assessing dairy products were included: Climate change, acidification, marine eutrophication, terrestrial eutrophication, freshwater eco-toxicity, and land use. Water resource depletion was not included due to lack of relevant foreground data, which is an important element for accurate assessments within this category. Although not included in the PEF guidelines, carbon sequestration was included in the climate change impact, to overcome the limitations of previous studies. For the same reasons, the effect on biodiversity based on Knudsen et al. (2017) was included. In order to capture possible differences in transport and use of N-fertilizer, the impact category 'Mineral, fossil and renewable resource depletion' was also included.

Allocation

A dairy system produces more than one product. Apart from milk (main product), two co-products are obtained: calves for meat and culled cows. Therefore, it was necessary to distribute the total environmental impact between the three products. The recommended allocation method in the PEF guide for dairy production (EC, 2015) was used in the present study, which is the same as suggested in the IDF (2015). The allocation factor is based on the ratio between fat and protein corrected milk (FPCM) and meat.

2.3.2 Life cycle inventory

Input-output data at farm level

Table 3 shows the inputs and outputs at farm level for the different dairy systems. The total feed requirement for milking cows and replacement heifers in terms of feed energy and protein was set according to Kristensen (2015) and Mogensen et al. (2015) across countries based on live-weight (requirement for maintenance), milk yield and live-weight gain. The intake of different feed items for cows and heifers was then estimated based on the land use, estimated crop yield and expert evaluations.

Estimation of climate impact of soil carbon changes

The impact of soil carbon changes were estimated according to the method suggested by Petersen et al. (2013). The method estimates climate impact of soil carbon changes based on the input of carbon (C) to the soil from above- and belowground crop residues and manure. The approach is based on one year's addition of carbon to the soil and the soil C dynamics of this carbon is modelled using a soil C model such as C-TOOL or RothC, to consider the soil C dynamics. Since part of this carbon will be emitted over a longer period, this modelling is combined with the Bern Carbon Cycle model to estimate the accumulated climate impact over time from that one year's addition of carbon to the soil. The soil C dynamics and the resulting effect on the atmospheric load of carbon by using the Bern Carbon Cycle model are not dependent on whether the soils has reached equilibrium or not. Furthermore, since the approach is designed for agricultural LCA's, it is focused on the effect of one crop on soil carbon sequestration – and not dependent on whether the practice will continue. By combining soil carbon modelling (in C-TOOL) and the Bern Carbon Cycle Model, Petersen et al. (2013) estimated for a sandy loam soil with

12% clay and a mean average air temperature of 8°C that 9.7% of the C added to the soil would be sequestered in a 100 years perspective. Since both Austria and UK has a mean average air temperature of 10°C, it is assumed that the 9.7% can be used for those countries as well. A 100 years time perspective was chosen due to the time perspective of the global warming potential. The actual amount of carbon in above and belowground crop residues were estimated based on coefficients from Taghizadeh-Toosi et al. (2014) and crop yields. The crop specific figures was based on Mogensen et al. (2014) that also used the Petersen et al. (2013) method to estimate soil carbon changes and also takes soil tillage into account. As the figures estimated by Mogensen et al. (2014) describe Danish conditions for crop production, they were adjusted in accordance with yields reported for the UK and Austria. The estimated C input to the soil from above and belowground crop residues were established for each feed crop. To estimate the soil carbon changes related to each feed crop, wheat was used as a reference crop to calibrate to the average C input to arable soils as described in Mogensen et al. (2014).

Estimation of on-farm emissions

Major on-farm emissions are methane from enteric fermentation and manure handling, nitrous oxide emissions from fields and manure, field emissions of nitrate, ammonia, phosphate and pesticides, and carbon dioxide related to use of fossil fuels. Since the N emissions are very important for several impact categories, and information about the internal N-turnover is needed for the estimation of several emissions, a nitrogen balance at farm and field level was established. This helped to ensure coherence in the assumed inputs and output as well as the internal turnover. N fixation was calculated as a fraction of the expected yield of legumes in relation to the added mineral N fertiliser (Kristensen et al., 2007). The crop specific figures for inputs used at crop level were established based on Mogensen et al. (2015). The approach used also led to estimation of the potential leaching (NO_3^--N or PO_4^{3-}-P) by deducting the losses (NH_3-N, NO-N, N_2O-N and N_2-N) and soil changes (N) from inputs and outputs (crop yields and straw).

Table 4 provides an overview of references used for the estimation of emissions.
N emissions were calculated at stable and field level and included several sources of pollutants: manure, mineral fertilizers and crop residues. Excreted manure was estimated according to Kristensen et al. (2005) by setting up an N balance at herd level. The emissions related to manure refer to emissions taking place at stable, at pasture, at storage and application. The emission factor was chosen in relation to the manure management system (slurry, solid manure, deep litter). The emissions regarding application of mineral fertilizer were calculated based on kg N applied. Crop residues were assumed those estimated by Mogensen et al. (2015) for Danish systems and small adjustments were made for the other systems according to the obtained crop yield. The emission factors were related to the N content in crop residues. P leaching potential was estimated at field level based on the P balance. It was assumed that 97% of the P surplus stays in soil and 3% is leached to waterbodies (Dalgaard et al., 2006). Table 5 shows the established N balances and the corresponding losses of nitrogenous substances.

Estimation of ecotoxicity impacts

Ecotoxicity impacts from pesticide use were evaluated following the LCIA emission-to-damage framework (Hauschild, 2005; Rosenbaum et al., 2015; Peña et al., 2018). Pesticide application practices in selected European dairy systems were investigated. It is assumed that crops are treated by foliar spray application and the agricultural field is considered as part of the ecosphere. Furthermore, 52 different pesticide a.i., from which, 28 were herbicides, 11 fungicides, 9 insecticides and 4 plant growth regulators, were assessed. Ecotoxicity impact scores were estimated by multiplying pesticide emission fractions with their respective characterization factors (Rosenbaum et al., 2008). The emission fractions linked to the use of pesticide a.i. are quantified using a static percentage distribution of a.i. into the different environmental compartments (air, water, crop and soil). This distribution represents fractions dependent on the spray application method and the drift functions (Balsari et al., 2007; Felsot et al., 2010; Gil et al., 2014). Characterization factors were calculated with USEtox 2.02 as characterization method, using the European landscape dataset (Fantke et al., 2015). Furthermore, for pesticide a.i. not included in USEtox database (i.e. mesotrione, foramsulfuron, epoxiconazole, pyraclostrobin, boscalid, fenopiridin, prothioconazole, metconazole and thiacloprid) the characterization factors from Peña et al. (2018) were used. The freshwater ecotoxicity impacts, representing the potential affected fraction of species (PAF m^3 d), is expressed in comparative toxic units (CTUe) per kg^{-1}FPCM. Finally, the total impact score for each agricultural system was calculated in an additive form.

Estimation of biodiversity impacts

Biodiversity impacts were estimated using the characterization factors suggested by Knudsen et al. (2017). The approach is using plant species as a proxy for biodiversity. The characterization factors is based on a unique data set derived from field recording of plant species diversity in farmland in six European countries within the *Temperate Deciduous Forest biome.* The effects on biodiversity were assessed as the potential reduction in biodiversity compared to natural conditions. Four overall land-use types managed as organic or conventional were considered and Table 6 provides the characterization factors used. For feed imported from outside the *Temperate Deciduous Forest* biome, a characterization factor of 0.68 PDF m^{-2} was used, which was the average of 0.76 in De Baan et al. (2013) and 0.60 in Mueller et al. (2014) and similar to the value found in Knudsen et al. (2017). Thus, 0.68 PDF m^{-2} was used for all conventional arable crops. Table 6 also provides for each system the land use of each of the four land use types per kg FPCM. The Biodiversity Damage Potential was calculated as the land use per kg milk (m^2 kg $FPCM^{-1}$) for a particular crop type multiplied with the characterization factor (PDF m^{-2}) and summed for all the crop types in the feed ration (Table 6).

Estimation of off-farm emissions

The Ecoinvent 3.3 database supplied through the SimaPro software by PRé Consultants was used for the assessment of country specific electricity, phosphate and potassium fertilizer and diesel and fuel oil and the Agrifootprint database was used for nitrogen fertilizer. With regard to the bought-in feed, it was assumed that the feedstuffs are produced at country level and only the soybean compounds are imported from other countries (Brazil for conventional systems and China for organic systems). The emissions associated with deforestation and land use change were excluded due to uncertainty. The feed production processes were established in SimaPro by using the processes described by Mogensen et al. (2015). For legumes, the processes were setup according to Knudsen et al. (2013). The transport distances were assumed the same in all systems for the feed produced at country level (168 km). This transport distance was estimated for Danish conditions (Mogensen et al. (2018) where much of the feed is produced as close as possible to the farmer. The imported soybean compounds were assumed to be transported by sea to Europe (The Netherlands) and then by train to the different countries.

3. Results and discussion

3.1 Environmental impact of organic and conventional milk

Figures 1, 2 and 3 show the overall environmental impact of organic and conventional milk production in the different systems for mixed (DK), grassland-based (UK) and mountainous systems (AT). It is clear that across production systems the environmental impact per kg fat and protein corrected milk (FPCM) for climate change, terrestrial eutrophication and acidification are similar or slightly lower for organic milk compared to conventional, whereas the impact on resource use, biodiversity and ecotoxicity is clearly lower under organic management (Figure 1-3).

Soil carbon changes

The soil carbon changes are included in the climate change category and reduces the carbon footprint by 5-18% (Figure 4). The lowest contribution from soil carbon changes was in the mixed conventional system and the highest was in the mountainous organic systems. The contribution from soil carbon changes are positively correlated to the amount of grass in the feed ration, since carbon sequestration is higher for grass than arable crops (Mogensen et al., 2014). For the same reason, organic systems have a higher carbon sequestration than conventional systems and the grass-based and mountainous systems have a higher C sequestration than the mixed systems (Figure 4). The carbon sequestration is here estimated in a 100 years perspective to comply with time perspective in the global warming potential. If a 20 years time perspective had been chosen, as in the IPCC, the sequestration factor would be twice as high according to Petersen et al. (2013) and thus also the carbon sequestration. The weakness of the soil carbon sequestration method suggested by Petersen et al. (2013) is that it lacks values for all pedo-climatic conditions globally, so the 9.7% can only be used for pedo-climatic condition comparable to Denmark. For UK and Austria, this assumption seems valid, but for the use in e.g. warmer climates, new percentages should be calculated. A challenge of the method is also that it requires an estimation of the carbon input to the soil from above and belowground crop residues, which requires an estimate of the above-(yields) and belowground biomass. Little research are still available on the root biomass, which can be seen in the IPCC guidelines and soil carbon models. However, this challenge is valid to all estimations and modelling of soil carbon and more research is needed on estimation of root biomass, especially in grasslands. To overcome those two challenges, carbon sequestration percentages should be calculated for more pedo-climatic zones and better estimates of especially root biomass should be available. Some of the advantages of using the Petersen et al. (2013) method for estimating soil carbon changes, as compared to the IPCC method or the static numbers used in Kristensen et al. (2011) and Guerci et al. (2013), is that it is based on the actual input of carbon to the soil (and not default categories or one value). Furthermore, it considers the soil emission dynamics over time by using soil dynamic models such as e.g. C-TOOL and it considers the accumulated climate impact of the time dependent emissions by combining the soil carbon models with the Bern Carbon Cycle model, which would not be taken into account if a soil model alone were used. Finally, the method can be applied to real farms, when the yield is known (above and belowground) and a carbon sequestration percentage is available.

Climate change, eutrophication, acidification, land use and resource depletion

Previous studies have mainly focused on climate change, eutrophication, acidification and land use (e.g. Cederberg & Mattsson, 2000; Hörtenhuber et al., 2010; Salvador et al., 2016 among others). Results have illustrated similar or slightly lower impacts per kg milk under organic management, although land-use generally increases. The results from this study concur with these prior assessments. In the present study, the global warming potential for milk ranged from 0.74 kg CO_2 eq. kg^{-1} FPCM in the organic grassland systems to 1.01 kg CO_2 eq. kg^{-1} FPCM in the conventional mixed systems (Figure 1-3). This is comparable to previous studies, where a carbon footprint of milk has also been around 1 kg CO_2 eq. kg^{-1} FPCM (e.g. Cederberg and Mattsson, 2000; van der Werf et al., 2009; Salou et al., 2016). Since soil carbon sequestration is included in this study, the values will be lower especially for organic milk (Table 2). With regard to climate change, terrestrial eutrophication and acidification, the impacts of organic milk ranged from 75-95% of conventional (Figure 1-3) across the three systems. The main reason for the slightly lower impact on terrestrial eutrophication and acidification in the organic systems can be ascribed to an almost 50% lower ammonia emission per ha from manure because of greater outdoor access (Table 5).

Marine eutrophication is lower than conventional in the organic mixed and grassland-based systems and almost the same in the organic and conventional mountainous systems. Since the main contributor to this impact category is N leaching from agricultural fields, and because a determining factor in the organic system is the estimated biological N_2 fixation, this impact category is highly uncertain. The land use in the present study was ranged from 1.0-2.0 m^2 kg^{-1} FPCM per kg (Figure 1-3), which is in agreement with previous studies (e.g. Thomassen et al., 2008; van der Werf et al., 2009, Salou et al., 2016). When including only the above-mentioned impact categories, the environmental impact of organic milk was similar or slightly lower compared to conventional milk, per kg of product. The impact on resource depletion for organic milk is on average 20% of the impact from conventional milk across the three systems (Fig. 1-3), which is mainly related to the avoidance of mineral fertilizer.

Biodiversity and ecotoxicity

When including the impact categories of ecotoxicity and biodiversity, the overall picture changes considerably. With regard to ecotoxicity, there was a clear difference between organic and conventional milk (Figure 1-3). On average, the impacts from organic milk was 2% of the conventional milk across the three systems. The impacts for conventional milk were in the range of 0.8-1.1 CTUe kg^{-1} FPCM, which is comparable to the impact of 1.1-1.5 CTUe kg^{-1} FPCM found in Chobtang et al. (2017), but much lower than the impacts of 213-544 CTUe kg^{-1} FPCM found in Salou et al. (2016). In the present study, the freshwater ecotoxicity impact for conventional milk was approximately 50 times higher than for organic milk, whereas Salou et al. (2016) reported an impact that was approximately 500 times higher for conventional compared to the organic milk. The inconsistencies in the ecotoxicity results presented here compared to results obtained by Salou et al. (2016) may be explained by the underlying assumptions for the inventory modelling (e.g., 100% emission to soil versus percentage distribution in environmental compartments) and the impact characterization (e.g., different versions of the characterization method). Even though these differences exist, the trends and patterns observed for ecotoxicity impact results were similar in both studies when expressed on a per hectare basis and across crops.

The biodiversity impact assessment also revealed clear differences between organic and conventional management. On average, the biodiversity impacts from the organic milk were 33% of the conventional across the three systems. In the grassland and mountainous systems, the values were negative for organic, compared to the mixed system, suggesting a potential increase in on-farm biodiversity. The biodiversity damage potential for the conventional milk was in the range of 0.26-0.48 PDF kg^{-1} FPCM, which is comparable to or slightly lower than the values found by Guerci et al. (2013) and Battini et al. (2016). Müller-Lindenlauf et al. (2010) did not ascribe a value per kg milk, but used a ranking scheme for the farm that did not include imported feed. Guerci et al. (2013) and Battini et al. (2016) used the approach and values suggested by De Schryver et al. (2010). The present study is also based on the approach suggested by De Schryver et al. (2010), but it uses the updated estimates for Europe as provided by Knudsen et al. (2017). The biodiversity value for conventional intensive arable crops are 0.79 PDF m^{-2} in De Schryver et al. (2010), but 0.68 PDF m^{-2} in Knudsen et al. (2017), which explains the slightly lower values in the present study. The biodiversity damage potential of organic milk was lower than conventional milk both due to the lower characterization factors for organic crops/grass in general (Table 6) and due to a higher proportion of grass in the feed rations for organic cows (Table 2).

One of the weaknesses of the characterization factors suggested by Knudsen et al. (2017) is that it is based on plant species diversity and captures only arthropods, birds etc. indirectly. However, it is a common approach to focus on plant species richness, which is also used by e.g. Mueller et al. (2014). Another applied approach is SALCA biodiversity as described by Jeanneret et al. (2014) and used by Nemecek et al. (2011). An expert system gives a score from 1-100% on the impact on a set of indicator species group. Thus, it is not using characterization factors as such and it is only valid for Swiss arable and grassland systems and adjacent regions. Furthermore, it cannot be reported along with the other LCA impacts since it uses a scale where high values are beneficial for environmental, which is the opposite in LCA impact categories. Another weakness of the Knudsen et al. (2017) characterisation factors is that they are only valid in parts of Europe. One the other hand, it is based on a unique dataset using the same sampling and estimation methods across the European countries involved. One of the major challenges with the biodiversity impact category in general is the availability of data and especially data on biodiversity impacts under organic and conventional management (Mueller et al., 2014), where the majority studies have been made in Europe. One of the strengths of the Knudsen et al. (2017) characterization factors is that they are able to distinguish between organic and conventional farm management practices contrary to e.g. Chaudhary et al. (2015). The characterization factors can be applied directly to real farms, while taking the farm management such as organic or conventional into account. It would be beneficial to develop the Knudsen et al. (2017) characterization factors further to consider more dimensions of biodiversity.

Effect of the type of dairy production system

Across the three types of systems, the contribution of enteric fermentation to climate change is similar or slightly higher in the organic systems, due to the lower milk yield per cow (Figure 4; Table 1). At the same time, the contribution of home-grown feed to climate change is similar or slightly lower in the organic system, due to the absence of artificial fertilizer. The avoidance of fertilizer in the organic system is also reflected in the contribution to resource depletion (Figure 5).

The mixed system is characterised by a higher yield per cow, more arable crops in the feed supply and a higher amount of bought-in concentrates than the grassland-based and mountainous system, independent of whether the production is organic or not. This is reflected in the resource depletion impact for the mixed conventional system, which is much higher than the other systems due to fossil energy-use related to imported feed, transport and the production of home-grown feed (Figure 5). Likewise, the organic production of milk from the grassland-based and mountainous systems contributes much less to this impact category than the organic milk from the mixed system.

The impact on ecotoxicity is mainly related to imported feed in the conventional mixed and grassland-based systems, whereas home-grown feed is the main contributor in the mountainous systems (Figure 6) due to the higher share of home-grown cereals in the feed supply. The highest impact is found in the conventional mixed systems.

Land use was approximately 50% higher for organic compared to conventional milk in the mixed and mountainous systems, whereas it was only 12% higher in the grassland-based system due to a combination of milk yield, grass yield and feed intake. The mixed organic systems has a higher impact on biodiversity than the two other systems, as pasture constitutes a much greater proportion of the land-use within the mountainous and grassland-based systems (Figure 7).

More grass instead of annual crops in the feed ration will reduce both the freshwater ecotoxicity and the biodiversity potential if the milk yield is not markedly affected. Thus, the study suggests that the main improvement options would be to increase the amount of grass in the ration and reduce the import of feed, which will affect both soil carbon sequestration in the climate change category, biodiversity and freshwater ecotoxicity.

5. Conclusions

The study illustrated that organic milk has a similar or slightly lower impact than conventional milk when considering the climate change, eutrophication and acidification categories, although land-use was found to be higher. Soil carbon changes reduced the global warming potential of milk by 5-18%. The impacts of organic milk production on freshwater ecotoxicity, biodiversity and resource depletion were 2, 33 and 20% of the impacts of conventional management, respectively, across the basic systems considered. Thus, the study highlights the importance of including more dimensions such as soil carbon sequestration, biodiversity and ecotoxicity in life cycle assessments of agricultural products, especially for organic or grass-based systems. The study shows that including more grass in the rations of dairy cows increases soil carbon sequestration and decreases the negative impact on biodiversity.

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References

AHDB Dairy, 2014. Greenhouse gas emissions on British dairy farms. DairyCo carbon footprinting study: year three, Stoneleigh Park, UK.

Baldini, C., Gardoni, D., Guarino, M., 2017. A critical review of the recent evolution of Life Cycle Assessment applied to milk production. J Clean. Prod. 140, 421-435.

Balsari, P., Marucco, P., Tamagnone M., 2007. A test bench for the classification of boom sprayersaccording to drift risk. Crop Protect. 26, 1482–1489.

Battini, F., Agostini, A., Boulamanti, A.K., Giuntoli, J., Amaducci, S., 2014. Mitigating the environmental impacts of milk production via anaerobic digestion of manure: case study of a dairy farm in the Po Valley. Sci. Total Environ. 481, 196-208.

Bengtsson, J., Ahnstroem, J.,Weibull, A.-C., 2005. The effects of organic agriculture on biodiversity and abundance: a meta-analysis. J. Appl. Ecology 42, 261–269.

Cederberg, C., Mattsson, B., 2000. Life cycle assessment of milk production – a comparison of conventional and organic farming. J. Clean Prod. 8, 49-60.

Chaudhary, A., Verones, F., de Baan, L., Hellweg, S., 2015. Quantifying land use impacts on biodiversity: combining species − area models and vulnerability indicators. Environ. Sci. Technol. 2015 (49), 9987–9995.

Chobtang, J., Ledgard, S.F., McLaren, S.J., Donaghy, DJ., 2017. Life cycle environmental impacts of high and low intensification pasture-based milk production systems: A case study of the Waikato region, New Zealand. J.Clean Prod. 140, 664-674.

Dalgaard, R., Schmidt, J., Halberg, N., Kristensen, I.S., 2006. Modelling representative and coherent Danish farm types based on farm accountancy data for use in environmental assessments. Agric. Ecosyst. Environ. 117, 223-237.

Dalgaard, R., Schmidt, J, Flysjö, A., 2014. Generic model for calculating carbon footprint of milk using four different life cycle assessment modelling approaches. J. Clean. Prod. 73:146-153.

Dammgen, U., Hutchings, N.J., 2008. Emissions of gaseous nitrogen species from manure management – a new approach. Environ. Pollut. 154, 488-497.

De Schryver, A.M., Goedkoop,M.J., Leuven, R.S.E.W., Huijbregts,M.A.J., 2010. Uncertainties in the application of the species area relationship for characterisation factors of land occupation in life cycle assessment. Int. J. Life Cycle Assess. 15, 682–691.

De Baan, L., Alkemade, R., Koellner, T., 2013. Land use impacts on biodiversity in LCA: a global approach. Int. J. Life Cycle Assess. 18 (6), 1216–1230.

EC, 2011a. A Roadmap for moving to a competitive low carbon economy in 2050. http://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX%3A52011DC0112

EC, 2011b. Our life insurance, our natural capital: an EU biodiversity strategy to 2020. http://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:52011DC024 4

EC, 2013. The environmental footprint pilots. http://ec.europa.eu/environment/eussd/smgp/ef_pilots.htm

EC, 2015. Product environmental footprint category rules for Dairy Products – draft August 2015.

EC, 2016. Proposal for an Effort Sharing Regulation 2021-2030. https://ec.europa.eu/clima/policies/effort/proposal_en

EC, 2018a. Product Environmental Footprint Category 2 Rules Guidance 3 Version 6.3, http://ec.europa.eu/environment/eussd/smgp/pdf/PEFCR_guidance_v6.3.pdf

EC, 2018b. Product Environmental Footprint Category Rules for Dairy Products http://ec.europa.eu/environment/eussd/smgp/pdf/PEFCR-DairyProducts_2018-04-25_V1.pdf

EEA, 2007. EMEP/CORINAIR Emission inventory guidebook – 3rd edition. 2007 Update. Technical report no. 30. Environmental European Agency, Copenhagen, Denmark, 2374 pp.

Felsot, A.S., Unsworth, J.B., Linders, J.B.H.J., Roberts, G., Rautman, D., Harris, C., Carazo, E., 2010. Agrochemical spray drift; assessment and mitigation—A review. J. Environ. Sci. Heal. Part B 46, 1–23.

Fantke, P., Huijbregts, M., Margni, M., Hauschild, M., Jolliet, O., McKone, T., Rosenbaum, R., Meent, D.v.d., 2015. USEtox® 2.0 User Manual (Version 2). UNEP/SETAC scientific consensus model for characterizing human toxicological and ecotoxicological impacts of chemical emissions in life cycle assessment. http://usetox.org

Fisher, G. 2013. Feed planning for cattle and sheep. Tried and Tested. Available online at: http://www.nutrientmanagement.org/assets/12028

Fliessbach, A., Oberholzer, H.-R., Gunst, L., Mäder, P., 2006. Soil organic matter and biological soil quality indicators after 21 years of organic and conventional farming. Agric. Ecosyst. Environ. 228, 273-284.

Fossum, J.P., 2014. Calculation of Carbon Footprint of Fertilizer Production. Note issued by Yara HESQ. http://yara.com/doc/122597_2013_Carbon_footprint-of_AN_Method_of_calculation.pdf.

Gattinger, A., Muller, A., Haeni, M., Skinner, C., Fliessbach, A., Buchmann, N., Mäder, P., Stolze, M., Smith, P., Scialabba, N.E.-H., Niggli, U., 2012. Enhanced top soil carbon stocks under organic farming. Proc. Natl. Acad. Sci. U.S.A. 109, 18226–18231.

Gil, Y., Balsari, P., Gallarta, M., Llorensc, J., Maruccob, P., Gummer Andersend, P., Fàbregasa, X., Llopa, J., 2014. Determination of drift potential of different flat fan nozzles on a boom sprayer using a test bench. Crop Protect. 56, 58-68.

Guerci, M., Knudsen, M.T., Bava, L., Zucali, M., Schönbach, P., Kristensen, T., 2013. Parameters affecting the environmental impact of a range of dairy farming systems in Denmark, Germany and Italy. J. Clean. Prod. 54, 133-141.

Gyldenkærne, S., Albrektsen, R., 2008. Revurdering af ammoniakemissionen 2003-2007. baggrundsnotat til vandmiljøplan III. Danmarks Miljøundersøgelser, Aarhus Universitet, Denmark. http://bios.au.dk/fileadmin/Resources/DMU/Vand/7-ammoniakemission.pdf

Hauschild, M (2005). Assessing environmental impacts in a life cycle perspective. Environ. Sci. Technol. 39 (4), 81A–88A.

Hietala S, Smith L, Knudsen MT, Kurppa S, Padel S, Hermansen JE. Carbon footprints of organic dairying in six European countries—real farm data analysis. Org. Agric. 5, 91-100.

Hole, D.G., Perkins, A.J.,Wilson, J.D., Alexander, I.H., Grice, P.V., Evans, A.D., 2005. Does organic farming benefit biodiversity? Biol. Cons. 122, 113–130.

Hörtenhuber, S., Lindenthal, T., Amon, B., Markut, T., Kirner, L., Zollitsch, W., 2010. Greenhouse gas emissions from selected Austrian dairy production systems-model calculations considering the effects of land use change. Renew. Agr. Food Syst. 25(4), 316-329.

IACS (2017) Integrated Administration and Control System (IACS). Online at https://ec.europa.eu/agriculture/direct-support/iacs_en

IDF, 2015. Bulletin of International Dairy Federation No. 479/2015. A common footprint approach for the dairy sector. The IDF guide to standard life cycle assessment methodology.

IPCC, 2006. 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Intergovernmental Panel of Climate Change (IPCC), National Greenhouse Gas Inventories Programme. http://www.ipcc-nggip.iges.or.jp/public/2006gl/index.html

Jeanneret, P., Baumgartner, D.U., Freiermuth Knuchel, R., Koch, B., Gaillard, G., 2014. An expert system for integrating biodiversity into agricultural life-cycle assessment. Ecol. Indic. 46, 224–231.

Knudsen, M.T., Topp, K., Olesen, J.E., Hermansen, J.E., Schelde, K., Angelopoulos, N., Reckling, M., 2013. Legumes future report D6.3. Energy and nutrient flows for the novel cropping systems – combined in a life cycle assessment of grain legumes.

Knudsen, M.T., Meyer-Aurich, A., Olesen, J.E., Chirinda, N., Hermansen, J.E., 2014. Carbon footprints of crops from organic and conventional arable crop rotations - using a life cycle assessment approach. J. Clean. Prod. 64, 609–618.

Knudsen MT, Hermansen JE, Cederberg C, Herzog F, Vale J, Jeanneret P, Sarthou J-P, Friedel, J.K., Balázs, K., Fjellstad,W., Kainz,M.,Wolfrum, S., Dennis, P., 2017, Characterization factors for land use impacts on biodiversity in life cycle assessment based on direct measures of plant species richness in European farmland in the 'temperate broadleaf and mixed forest' biome. Sci. Total Environ. 580, 358–366.

Kristensen, T., 2015. Beregning af grovfoderudbytte på kvægbrug ud fra regnskabsdata. DCA Rapport no. 057, March 2015, Aarhus University.

Kristensen, I.S., Halberg, N., Nielsen, A.H., Dalgaard, R., 2005. N-turnover on Danish mixed farms. Part II, in Bos, J., Pflimlin, A., Aarts, F., Vertes, F., (Eds), Nutrient management on farm scale. How to attain policy objectives in regions with intensive dairy farming. Report of the first workshop of the EGF Workshop. http://agrsci.dk/var/agrsci/storage/original/application/21bf0ab8f732a82cffed8951921067d6.pdf

Kristensen, I. S., Kristensen, T., Vinther, F. P. og Høgh-Jensen, H., 2007. Afgrødernes kvælstoffiksering. Omfang og metoder til beregning. Udkast til rapport ved Ib Sillebak Kristensen.

Kristensen, T., Mogensen, L., Knudsen, M.T., Hermansen, J.E., 2011. Effect of production system and farming strategy on greenhouse gas emissions from commercial dairy farms in a life cycle approach. Livest. Sci. 140, 136-148.

Lampkin, N., Measures, M., Padel, S., 2012. 2011/12 Organic Farm Management Handbook. The Organic Research Centre, Elm Farm, Newbury, UK.

Landsbruginfo, 2015. Landbrugsinfo.dk. Online at: https://landbrugsinfo.dk/kvæg/Tal-om-kvæg/Sider/pif003tabel20.aspx .

Meier, M.S., Stoessel, F., Jungbluth, N., Juraske, R., Schader, C., Stolze, M., 2015. Environmental impacts of organic and conventional agricultural products–Are the differences captured by life cycle assessment? Journal of Environmental Management 149, 193-208.

Mikkelsen, M.H., Gyldenkærne, S., Poulsen, H.D., Olesen, J.E., Sommer, S.G., 2006. Emission of ammonia, nitrous oxide and methane from Danish Agriculture 1985-2002. Methodology and Estimates. National Environmental Research Institute, Denmark. 99 pp – Research Notes from NERI No. 231. http://www.dmu.dk/Pub/AR231.pdf

Mikkelsen, M.H., Gyldenkærne, S., Poulsen, H.D., Olesen, J.E., Sommer, S.G , 2005. Estimates and methodology for emissionof ammonia and greenhouse gasses from Danish Agriculture 1985-2002. National Environmental Research Institute, Denmark. Working report from NERI No. 204. (In Danish with English summary). 83 pp. http://www2.dmu.dk/Pub/AR231.pdf

Moakes, S., Lampkin., N., Gerrard, C., 2015. Organic farms incomes in England and Wales 2013/14. Newbury/Aberystwyth Organic Research Centre/Organic Centre Wales Aberystwyth University.

Mogensen, L., Kristensen, T., Nguyen, T.L.T., Knudsen, M.T., Hermansen, J.E., 2014. Method for calculating carbon footprint of cattle feed – including contribution from soil carbon changes and use of cattle manure. J. Clean. Prod. 73, 40-51.

Mogensen, L., Hermansen, J.E., Nguyen, T.L., Preda, T., 2015. Environmental assessment of beef by life cycle assessment – 13 danish beef production systems. DCA Report no. 061, April 2015, Aarhus University.

Mogensen, L., Knudsen, M.T., Dorca-Preda, T., Nielsen, N.I., Kristensen, I.S., Kristensen, T. (2018) Bæredygtighedsparametre for konventionelle fodermidler til kvæg – metode og tabelværdier (in Danish) (Sustainability parameters for conventional feed for cattle – methods and numbers). DCA report No. 116. 129 p. Online at http://web.agrsci.dk/djfpublikation/index.asp?action=show&id=1260

Mueller, C., de Baan, L., Koellner, T., 2014. Comparing direct land use impacts on biodiversity of conventional and organic milk - based on a Swedish case study. Int. J. Life Cycle Assess. 19 (1), 52–68.

Müller-Lindenlauf, M., Deittert, C., Köpke, U., 2010. Assessment of environmental effects, animal welfare and milk quality among organic dairy farms. Livestock Science 128, 140-148.

NaturErhvervstyrelsen (2015) Vejledning om gødsknings-og harmoniregler, 2015. Planperioden 1. august 2015 til 31. juli 2016. Miljø – og Fødevareministeriet, NaturErhvervstyrelsen. http://lbst.dk/fileadmin/user_upload/NaturErhverv/Filer/Landbrug/Goedningsregnskab/Vejledning_om_goedsknings-_og_harmoniregler_nyeste1.pdf

Nemecek, T., Kagi, T., 2007. Life Cycle Inventories of Agricultural Production Systems. Final Report ecoinvent v2.0 No. 15, Swiss Centre for Life Cycle Inventories, Duebendorf, CH.

Nemecek, T., Dubois, D., Huguenin-Elie, O., Gaillard, G., 2011a. Life cycle assessment of Swiss farming systems: I. Integrated and organic farming. Agric. Syst. 104: 217-232.

Nielsen, O.K., Plejdrup, M.S., Winter, M., Nielsen, M, Gyldenkærne, S., Mikkelsen, M.H., Albrektsen, R., Thompsen, M., Hjelgaard, K., Hoffmann, L., Fauser, P., Bruun, H.G., Johannsen, V.K., Nord-Lassen, T., Vesteradal, L., Møller, I.S., Caspersen, O.H., Rasmussen, E., Petersen, S.B., Baunbæk, L., Hansen, M.G., 2013. Denmark's National Inventory report 2013. Emission Inventories 1990-2011 – Submitted under the United Nations Framework Convention on Climate Change and the Kyoto Protocol. DCE Report no. 56.

Ørum JE, Samsøe-Petersen L, 2014. Bekæmpelsesmiddelstatistik 2013: behandlingshyppighed og belastning. http://www2.mst.dk/Udgiv/publikationer/2014/12/978-87-93283-33-6.pdf

Parajuli, R., Kristensen, I.S., Knudsen, M.T., Mogensen, L., Corona, A., Birkved, M., Dalgaard, T., 2017. Environmental life cycle assessments of producing maize, grass-clover, ryegrass and winter wheat straw for biorefinery. J. Clean. Prod. 142(4), 3859–3871.

Peña, N., Knudsen, M.T., Fantke, P., Anton, A., Hermansen, J.E., 2018. Freshwater ecotoxicity assessment of pesticide use in crop production: Testing the influence of modeling choices. Manuscript submitted for publication.

Petersen, B.M., Knudsen, M.T., Hermansen, J.E., Halberg, N., 2013. An approach to include soil carbon changes in the life cycle assessments. J. Clean. Prod. 52, 217-224.

Ritchie, H., Reay, D.S., Higgins, P. (2018) The impact of global dietary guidelines on climate change. Global Envir. Change 49: 46-55.

Rosenbaum RK, Bachmann TMK, Gold LS, Huijbregts MAJ, Jolliet O,Juraske R, Koehler A, Larsen HF,MacLeod M,Margni M, McKone, TE, Payet J, Schuhmacher M, Van de Meent D, Hauschild MZ (2008) USEtox—the UNEP/SETAC-consensus model: recommended characterisation factors for human toxicity and freshwater ecotoxicity in life cycle impact assessment. Int J Life Cycle Assess 13: 532–546

Rosenbaum, R., Anton, A., Bengoa, X., Bjørn, A., Brain, R., Bulle, C., Cosme, N., Dijkman, T.J., et al. 2015. The Glasgow consensus on the delineation between pesticide emission inventory and impact assessment for LCA. The International Journal of Life Cycle Assessment, 20, 765-776.

Salou, T., Le Mouel, C., van der Werf, Hayo M.G., 2017. Environmental impacts of dairy system intensification: the functional unit matters! J. Clean. Prod 140, 445-454.

Salvador, S., Corazzin, M., Piasentier, E., Bovolenta, S., 2016. Environmental assessment of small-scale dairy farms with multifunctionality in mountain areas. J. Clean. Prod. 124, 94-102.

Statistics Denmark (2015). Statistics Denmark. Online at http://www.dst.dk/en/Statistik#

Taghizadeh-Toosi, A., Christensen, B. T., Hutchings, N. J., Vejlin, J., Kätterer, T., Glendining, M. og Olesen, J. E., 2014. C-TOOL – A soil carbon model and its parameterisation. Ecological modelling 292: 11-25.

The Professional Nutrient Management Group, 2015. Feed Planning for Cattle and Sheep Guide. Catchment Sensitive Farming Series, Stoneleigh Park, UK. Online at http://www.nutrientmanagement.org/home/

Thoma, G. J., J. S. Popp, D. W. Nutter, D. R. Shonnard, R. Ulrich, M. D. Matlock, D. Kim, Z. Neiderman, N. Kemper, C. East, Adom, F., 2013. Greenhouse gas emissions from milk production and consumption in the United States: A cradle-to-grave life cycle assessment circa 2008. Int. Dairy J. 31: S3–S14

Thomassen, M.A., van Calker, K.J., Smits, M.C.J., Iepema, G.L., de Boer, I.J.M., 2008. Life cycle assessment of conventional and organic milk production in the Netherlands. Agric. Syst. 96, 95-107.

Tuck, S.L.,Winqvist, C., Mota, F., Ahnstrom, J., Turnbull, L.A., Bengtsson, J., 2014. Land-use intensity and the effects of organic farming on biodiversity: a hierarchical meta-analysis. J. Appl. Ecol. 51 (3), 746–755.

Van der Werf, H.M.G., Kanyarushoki, C., Corson, M.S., 2009. An operational method for the evaluation of resource use and environmental impacts of dairy farms by life cycle assessment. J. Environ. Manage 90, 3643-3652.

Vinther, F.P, 2005. SIMDEN – A simple model for quantifying denitrification and N2O emission. In: Stensberg, M., Nilsson, H., Brynjolfsson, R., Kapuinen, P., Morken, J., Birkmose, T.S. (eds.). Manure – an agronomic and environmental challenge. NJF seminar no. 372, 5-6 September 2005, Nils Holgerssongymnasiet, Skurup, Sweden.

Weidema, B., Wesnæs, M., Hermansen, JE:, Kristensen, T., Halberg N. 2008. Environmental improvement potentials of meat and dairy products. JRC report. ISBN 978-92-79-09716-4 , 194 pp

Tables

Table 1. Overview of the dairy production systems considered.

	Mixed (Denmark)		Grassland-based (UK)		Mountainous (Austria)	
	Conv.	Organic	Conv.	Organic	Conv.	Org.
Herd						
Breed	Holstein Friesian		Holstein Friesian*		Brown Swiss/ Simmental	
Milking cows (heads year^{-1})	168	168	118	127	12	10
Average weight (kg cow^{-1})	584	584	625	575	650	650
Young stock (heads cow^{-1})	0.91	0.91	0.58	0.58	0.58	0.58
Culling rate (%)	42	42	25	25	23	22
Production (cow-year^{-1})						
Sold milk (kg FPCM)	9599	8708	7411	6193	6230	5500
Sold calves (no.)	0.64	0.64	0.72	0.72	0.75	0.82
Culled cows, kg live-weight	245	245	157	144	152	146
Days at pasture (days animal^{-1} year^{-1})						
Milking cows	0	150	172	224	20	80
Young stock	150	150	149	224	70	110
Farm area (ha cow^{-1})	**0.88**	**1.21**	**1.09**	**1.20**	**0.96**	**1.28**
Cereals for feed (%)	15	20	11	7	23	12
Cereals for sale (%)	8					
Legumes (%)				1		4
Forage crops:						
maize silage (%)	33	5	9		16	
whole crop barley (%)	5	16				
temporary pasture*** (%)	38	59	41	53	12	11
permanent pasture (%)			39	39	50	74
Manure on crops (kg N ha^{-1})	170	129	104	104	121	88
Mineral N fertilizer (kg N ha^{-1})	58		134		49	
Bought in concentrates (kg DM ha^{-1})	2437	1321	1422	695	535	470
Pesticides (g a.i. ha^{-1})	496	0	239	0	385	0
Crop yields (kg DM ha^{-1})						
Cereals	5081	3511	5935	3088	5557	3550
Legumes				2295		1620
Grass clover/grass/Lucerne	7667	7211	7642**	7275**	7875**	6300**
Maize silage	11296	7732	12000**		10837**	
Whole crop cereals	7996	6121				
Permanent pasture			2213**	2213**	5814**	4750**

* In organic systems, there is also NZ Friesian, British Friesian and cross breeding (including Swedish Red and beef breeds)
** The yields for the forage crops refer to a utilization of 75% (Fisher, 2013, Table 16). UK permanent pasture yields are for grazed zero-input pasture.
*** Grassland less than 5 years of age, included in a crop rotation.

Table 2. Feed intake per year-cow in the dairy systems.

	Mixed (Denmark)		Grassland-based (UK)		Mountainous (Austria)	
	Conv.	Organic	Conv.	Organic	Conv.	Organic
Total feed intake (kg DM animal^{-1} year^{-1})	7333	7058	6272	5791	5674	5490
Concentrates						
Cereals, home-grown (%)	11	12	10	5	20	10
Imported soymeal/cake (%)	5	2			2	
Bought-in other concentrates (%)	21	21	24	14	7	11
Roughage						
Grazing (%)		14	14	21	3	12
Grass/clover/lucerne silage or hay (%)	27	29	39	60	44	67
Maize silage (%)	31	7	13		24	
Other (%)	5[1]	14[1]				

[1]Barley whole crop silage

Table 3. Input-output at farm level of the different dairy production systems per ha per year.

	Mixed (Denmark)		Grassland-based (UK)		Mountainous (Austria)	
	Conv.	Organic	Conv.	Organic	Conv.	Org.
Farm area (ha)	148	203	129	153	11.5	12.8
Input						
Feed (kg DM ha^{-1})	2437	1321	1422	695	535	470
Fertilizer (kg N ha^{-1})	58		134		49	
Diesel (litres ha^{-1})	92	64	57	42	84	14
Electricity (kWh ha^{-1})	900	697	720	696	853	722
Herbicides (g ha^{-1})	145	0	135	0	165	0
Plant growth regulator (g ha^{-1})	19	0	21	0	21	0
Insecticides (g ha^{-1})	28	0	29	0	30	0
Fungicides (g ha^{-1})	6	0	9	0	10	0
Output						
Cereals (kg DM ha^{-1})	135					
Milk (kg FPCM ha^{-1})	10751	7033	6779	5141	6501	4297
Calves (kg live weight ha^{-1})	29	21	26	19	24	16
Culled cows (kg live weight ha^{-1})	465	335	143	119	158	102

Table 4. Emitted substances/consumed resources, sources of emissions and models used.

Emitted substance/Used resource	Source of emissions	Model/source
Methane (CH_4)	Enteric fermentation	IPCC (2006) Tier 2
	Animal excretion	IPCC (2006) Tier 2
	-calculation of animal excretion	Kristensen et al. (2005)
	-emission factors	Mikkelsen et al. (2006); Mikkelsen et al. (2005); Nielsen et al. (2013); IPCC (2006)
Nitrous oxide (N_20-N)	Animal excretion (stable, storage, application, pasture)	
	-calculation of excreted N	Kristensen et al. (2005)
	-emission factors	IPCC (2006)
	Mineral fertilizers	IPCC (2006)
	Crop residues	IPCC (2006)
	Indirect emissions (from NH_3, $N0_3$)	IPCC (2006)
Ammonia (NH_3-N)	Animal excretion (stable, storage, application, pasture)	
	-calculation of excreted N	Kristensen et al. (2005)
	-emission factors	Mikkelsen et al. (2006); Mikkelsen et al. (2005)
	Mineral fertilizers	Mikkelsen et al. (2006); Mikkelsen et al. (2005)
	Crop residues	Gyldenkærne & Albrektsen (2008)
Nitric oxide (NO-N)	Animal excretion	
	-calculation of excreted N	Kristensen et al. (2005)
	-emission factors stable & storage	Dammgen & Hutchings (2008)
	-emission factors application	Nemecek & Kagi (2007)
	-emission factors excreted at pasture	Nemecek & Kagi (2007)
	Mineral fertilizers	EEA (2007)
	Crop residues	Dammgen & Hutchings (2008)
Nitrogen (N_2-N)	Animal excretion (stable, storage, application, pasture)	
	-calculation of excreted N	Kristensen et al. (2005)
	-emission factors	Vinther (2005)
	Mineral fertilizers	Vinther (2005)
	Crop residues	Vinther (2005)
Nitrate (NO_3-N)	Leaching: annual crops and grassland	Kristensen et al. (2005)
Phosphate (PO_3^4-P)	Leaching: annual crops and grassland	Kristensen et al. (2005); Dalgaard et al. (2006)
Soil carbon (CO_2)	Soil carbon changes	Petersen et al. (2013)
Pesticides	Pesticide application	USEtox, Rosenbaum et al. (2008)
Biodiversity	Land occupation	Knudsen et al. (2017)

Table 5. Nitrogen balances and losses in the different dairy systems (kg N ha^{-1}).

INPUT/OUTPUT	Mixed (Denmark)		Grassland-based (UK)		Mountainous (Austria)	
	Conv.	Org.	Conv.	Org.	Conv	Org.
INPUTS						
Imported feed	125	43	26	13	35	23
Straw bedding						
Seeds	1	1		1	1	1
Biological fixation	24	73		97	40	57
Deposition	15	15	15	15	15	15
Mineral fertilizer	58		134	0	49	
TOTAL INPUT	223	131	175	125	140	96
OUTPUTS						
Cash crops	12					
Milk	56	36	35	27	34	22
Calves	0.7	0.5	0.7	0.5	0.6	0.4
Culled cows	7	5	4	3	4	3
Straw			2			2
TOTAL OUTPUT	76	42	41	30	38	27
BALANCE	**147**	**89**	**134**	**94**	**101**	**69**
LOSSES						
N_2O-N	4.3	2.8	3.8	2.6	3.3	2.3
NH^3-N						
stable and storage	21.1	12.1	12.0	10.3	24.2	12.8
Grazing	0.9	2.4	1.2	2.2	0.6	1.5
spreading and crops	21.8	5.7	13.6	10.0	11.5	7.0
N2-N						
stable and storage	4.1	2.3	2.3	1.9	2.1	0.7
Grazing	0.8	2.1	1.0	1.7	0.5	4.6
spreading and crops	7.6	4.1	8.1	3.9	5.7	3,6
NO-N						
stable and storage	1.4	0.8	0.8	0.6	9.2	5.3
Grazing	0,0	0.1	0.0	0.1	0	0.1
spreading and crops	0.5	0.1	1.1	0.1	0.5	0.2
indirect denitrification						
Ammonia	0.4	0.2	0.3	0.2	0.2	0.2
SOIL sequestration	11	16	9	12	6	4
LEACHING	**74**	**40**	**81**	**49**	**30**	**27**

Table 6. Biodiversity characterization factors in PDF (potentially disappeared fraction) (based on Knudsen et al. 2017), land use per kg FPCM and Biodiversity Damage Potential for organic and conventional milk in the three systems.

	Characterization factors		Mixed systems - DK		Grass based systems- UK		Mountainous systems - Austria	
	PDF pr. m^2		LU(m^2 kg $FPCM^{-1}$)					
	CONV	ORG	CONV	ORG	CONV	ORG	CONV	ORG
Annual crops[1]	0.68	0.29	0.67	0.84	0.63	0.42	0.57	0.59
Grass-clover, in rotation[1]	0.09	-0.12	0.32	0.69	0.06	0.85	0.08	0.55
Grass, in rotation[1]	0.12	-0.06			0.48		0.08	0.22
Permanent pastures[2]	-0.23	-0.34			0.49	0.64	0.63	0.63
Total LU (m^2 kg $FPCM^{-1}$)			0.99	1.53	1.66	1.91	1.36	1.99
Biodiversity Damage Potential (PDF kg $FPCM^{-1}$)			0.48	0.16	0.37	-0.20	0.26	-0.12

[1] Based on an average of German and Austrian numbers in Knudsen et al. (2017) (p.363 and Table 6) for more intensive agriculture.
[2] Based on an average of monocot and mixed pastures in all countries from Table 7 in Knudsen et al. (2017) to represent less-intensive permanent pastures.

Figures

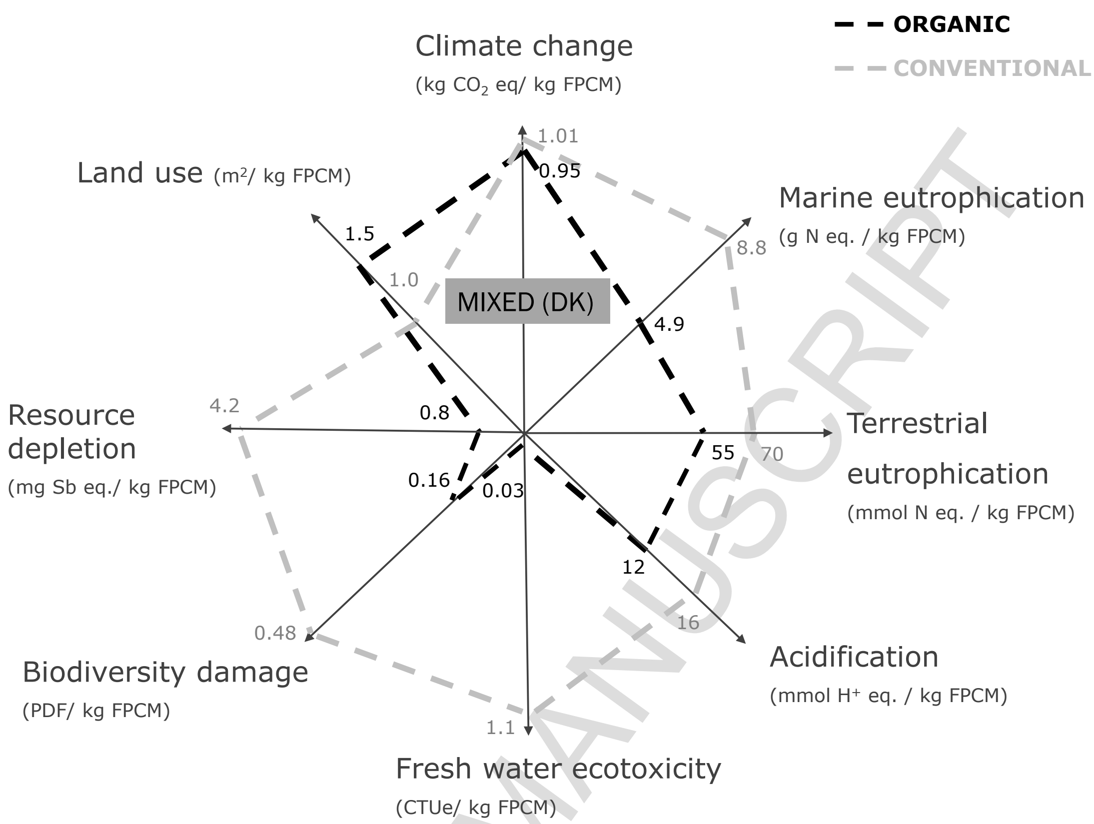

Figure 1. Environmental impact per kg fat and protein corrected milk (FPCM) in mixed dairy systems (Denmark).

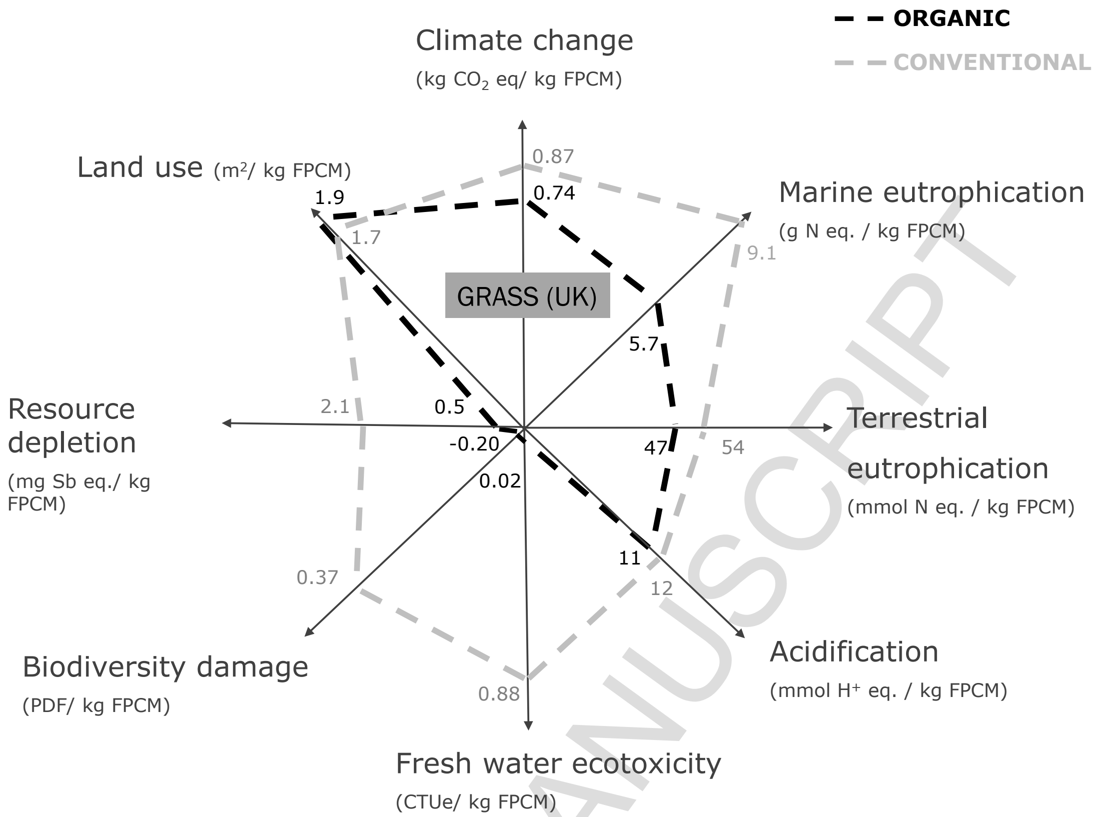

Figure 2. Environmental impact per kg fat and protein corrected milk (FPCM) in grassland-based dairy systems (UK).

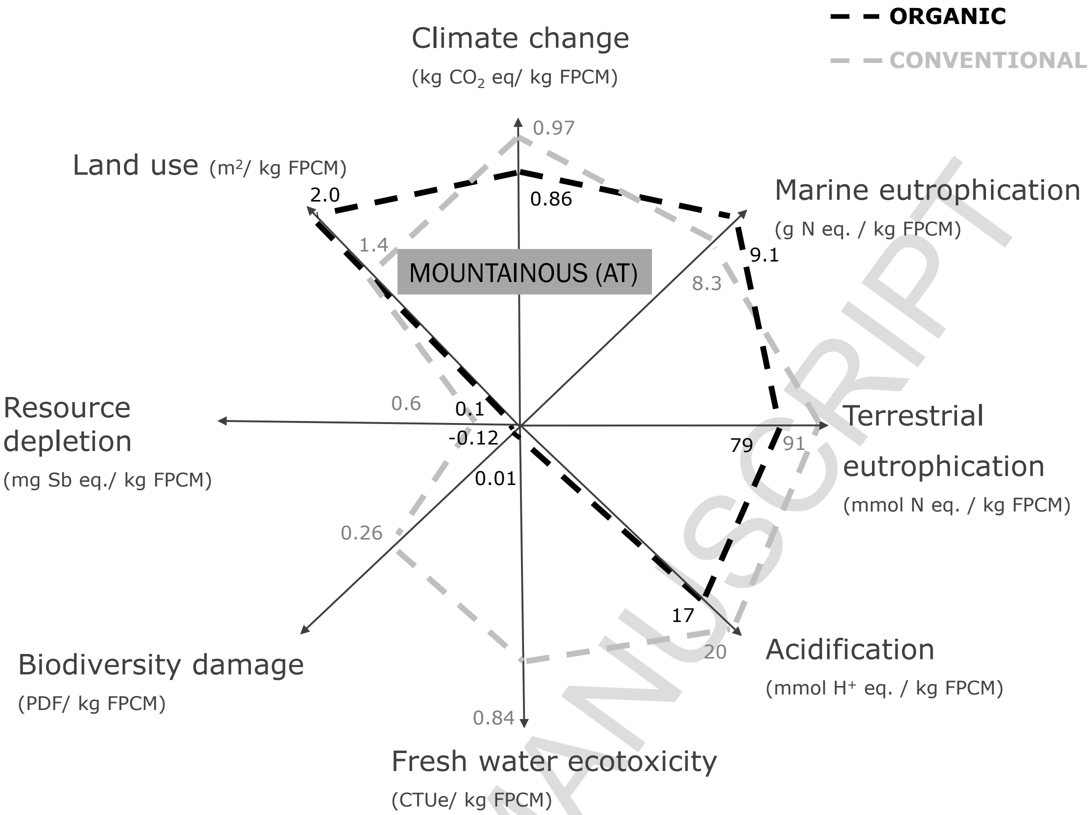

Figure 3. Environmental impact per kg fat and protein corrected milk (FPCM) in mountainous dairy systems (Austria).

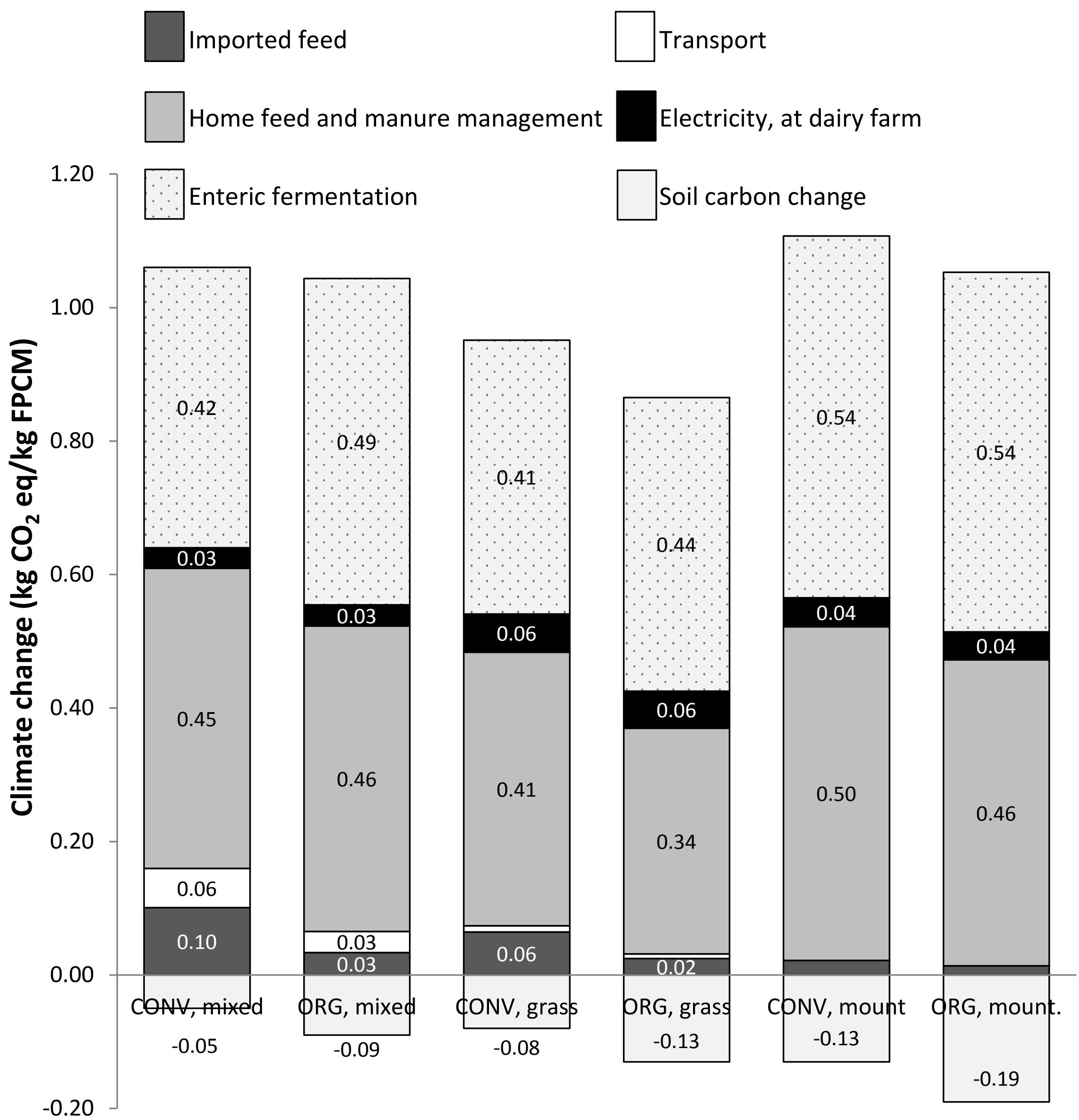

Figure 4. Contributions to climate change for organic (ORG) and conventional (CONV) fat and protein corrected milk (FPCM) produced in either mixed, grassland-based (grass) or mountainous (mount.) dairy systems.

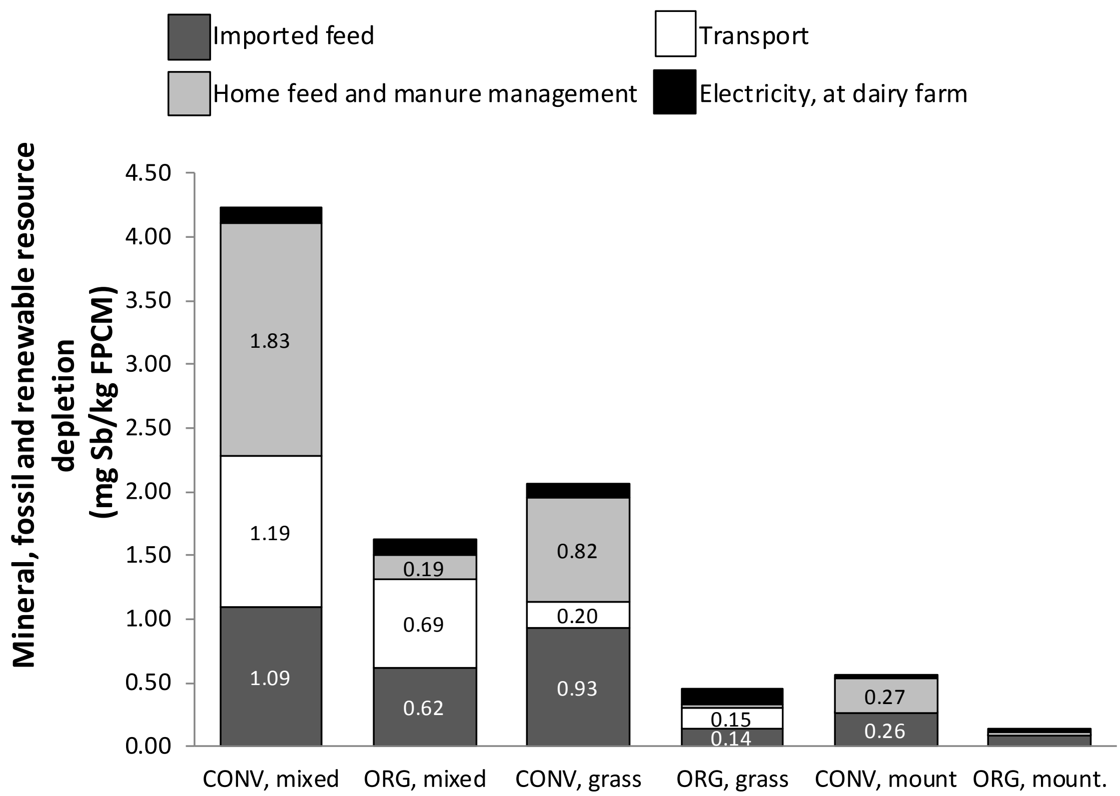

Figure 5. Contributions to mineral, fossil and renewable resource depletion for organic (ORG) and conventional (CONV) fat and protein corrected milk (FPCM) produced in either mixed, grassland-based (grass) or mountainous (mount.) dairy systems.

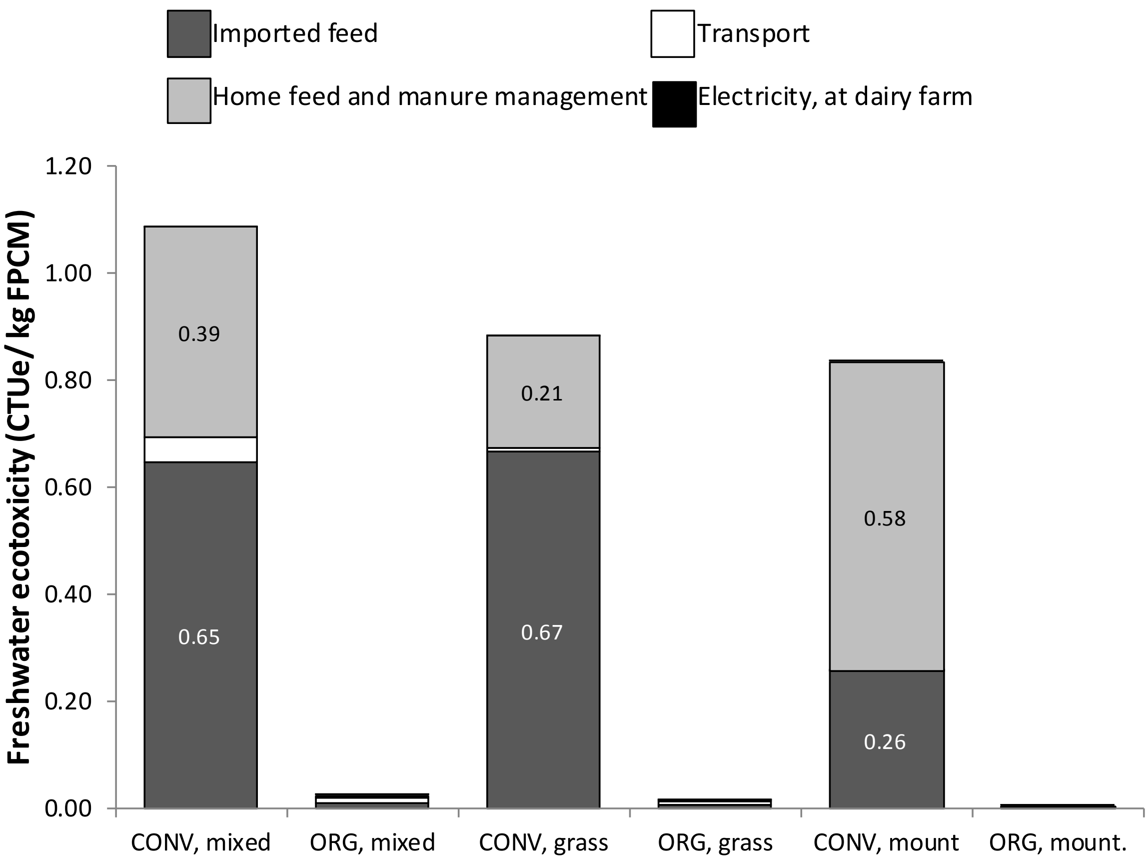

Figure 6. Contributions to freshwater ecotoxicity for organic (ORG) and conventional (CONV) fat and protein corrected milk (FPCM) produced in either mixed, grassland-based (grass) or mountainous (mount.) dairy systems.

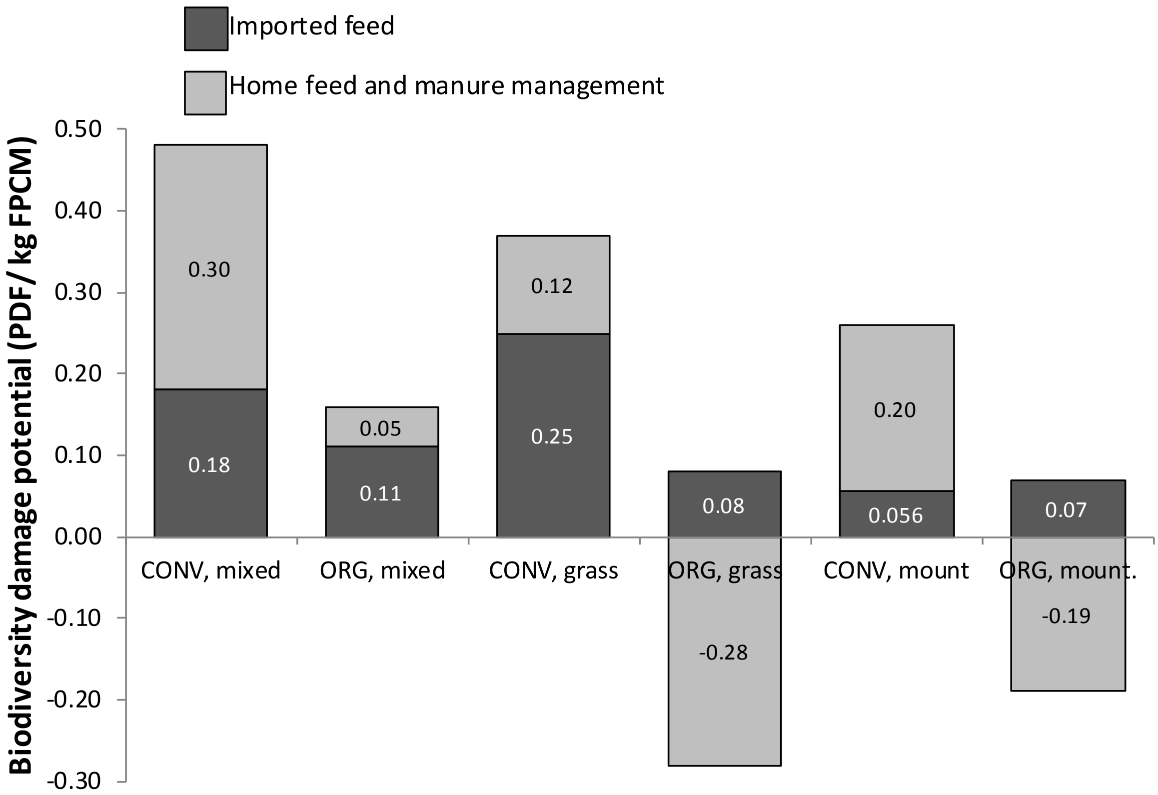

Figure 7. Contributions to biodiversity damage potential for organic (ORG) and conventional (CONV) fat and protein corrected milk (FPCM) produced in either mixed, grassland-based (grass) or mountainous (mount.) dairy systems.

The importance of including soil carbon changes, ecotoxicity and biodiversity impacts in environmental life cycle assessments of organic and conventional milk in Western Europe

Highlights

- The importance of including soil carbon changes, ecotoxicity and biodiversity in LCA is highlighted
- Including soil carbon changes reduced global warming potential of milk by 5-18%
- For ecotoxicity, organic milk had only 2% of the impacts of conventional milk
- Impacts on biodiversity of organic milk was only 33% of the conventional milk impacts
- Including more grass decreases impacts on biodiversity and increases soil C sequestration