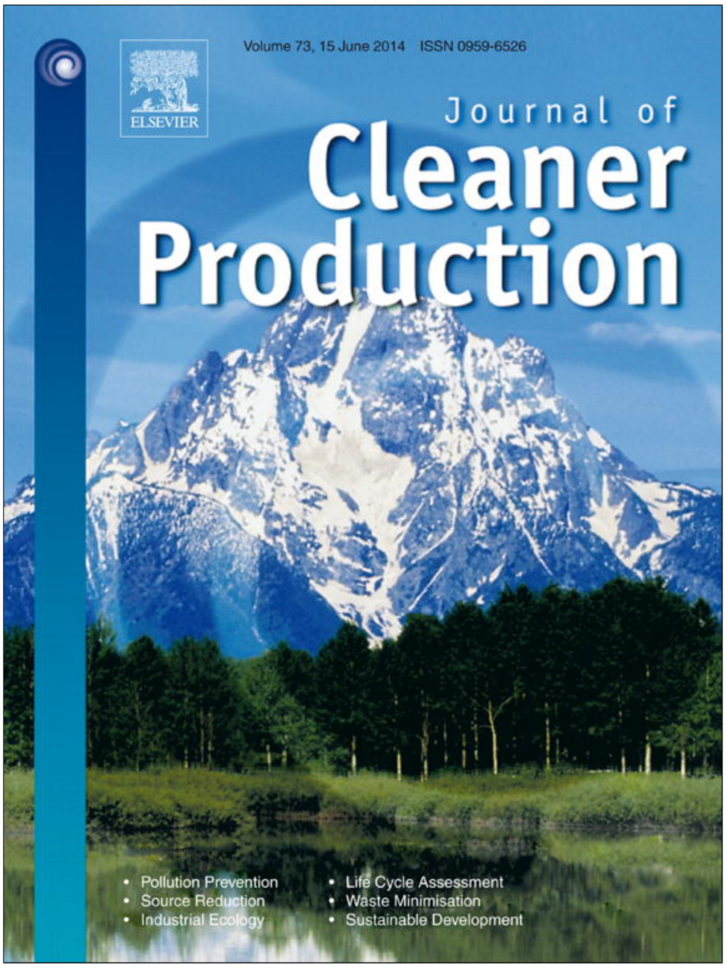
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# Method for calculating carbon footprint of cattle feeds – including contribution from soil carbon changes and use of cattle manure

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A B S T R A C T

Greenhouse gas emissions (GHG) related to feed production is one of the hotspots in livestock production. The aim of this paper was to estimate the carbon footprint of different feedstuffs for dairy cattle using life cycle assessment (LCA). The functional unit was '1 kg dry matter (DM) of feed ready to feed'. Included in the study were fodder crops that are grown in Denmark and typically used on Danish cattle farms. The contributions from the growing, processing and transport of feedstuffs were included, as were the changes in soil carbon (soil C) and from land use change (LUC). For each fodder crop, an individual production scheme was set up as the basis for calculating the carbon footprint (CF). In the calculations, all fodder crops were fertilized by artificial fertilizer based on the assumption that the environmental burden of using manure is related to the livestock production. However, the livestock system is also credited for the fact that the use of manure reduces the amount of artificial fertilizer being used. Consequently, a manure handling system was set up as a subsystem to the cattle system. This method allowed a comparison between different fodder crops on an equal basis. Furthermore, the crop-specific contribution from changes in soil C was estimated based on estimated amounts of C input to the soil.



## 1. Introduction

Livestock production is the world's largest user of land resources, with pasture and land dedicated to the production of animal feed representing almost 80% of the total agricultural area (FAO, 2010a). Thus, the production of animal feed can be considered as one of the major hotspots in the environmental impact from livestock production. For monogastric animals, Nguyen et al. (2010a) found that 64% of greenhouse gas (GHG) emissions was caused by feed production. In milk production, methane (mainly from enteric fermentation) makes the highest single contribution to GHG emissions, accounting for 50% or more of emissions on a global scale (FAO, 2010b), whereas nitrous oxide and carbon dioxide emissions related to feed production range from 27 to 38% and 5–10% of total emissions, respectively (FAO, 2010b). Thus, Kristensen et al. (2011) found that in Denmark up to 43% of the GHG emissions from milk production was related to feed production and manure handling and Flysjø et al. (2011) found that 38% of the emissions from milk production in New Zealand and 53% of the emissions from milk production in Sweden was related to feed production and manure handling. In beef production, Nguyen et al. (2010b) found that up to 55% of the GHG emissions from producing 1 kg beef meat was related to feed production.

The GHG emission from animal feed production comes from both the primary stage of crop production – primarily as $N_2O$ and from fossil energy related to fertilizer production – and from use of fossil energy in the processing of the crop into animal feed. The magnitude of the contribution of transport to the overall environmental impacts of animal feed varies, depending on whether the feedstuff is home-grown or imported. For example, slightly over 50% of total GHG emissions of soybeans imported from China to Denmark came from transport (Knudsen et al., 2010), whereas for locally produced roughage only between 0 and 13% of total GHG was due to transport (Vellinga et al., 2013).

In addition to these 'direct' effects, crop production also influences soil carbon sequestration depending on crop type and management (IPCC, 2006). Typically, grasslands are supposed to act as carbon sinks, whereas croplands release carbon (e.g. Vleeshouwers and Verhagen, 2002; Vellinga et al., 2004). Thus, a fair comparison between different fodder crops should ideally include such effects. However, so far, very few life cycle assessments (LCA) have included soil C sequestration in the overall GHG

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estimations, mainly due to methodological limitations. In the studies that do include soil C sequestration, the time horizon used is often less than the 100 years typically used for other emissions in an LCA (Gabrielle and Gagnaire, 2008; Hillier et al., 2009). Petersen et al. (2013) suggested how soil carbon changes could be included in LCAs by calculating a partial carbon budget for individual crops and combining it with the degradation and emissions of $CO_2$ from the soil and the resulting change in the atmosphere.

Around 12.2% of global GHG emissions in 2005 came from land use change (LUC), on par with the 13.8% originating from agricultural production (Herzog, 2009). As livestock is the world's largest user of land resources for feed production, LUC may contribute significantly to the GHG emission of animal feed. The question is how to account for this contribution. Basically, there are now two quite opposing approaches: a product-based and a land-based approach (Cederberg et al., 2013). According to the product-based approach, LUC is associated with the feeds grown in the regions where deforestation takes place (BSI, 2011), whereas in the land-based approach, LUC is a factor assigned to all feeds based on the assumption that all use of land for crop production increases pressure on land use, thus causing LUC somewhere in the world (e.g. Audsley et al., 2009).

Another unsolved question is how to account for the burden of using manure. Different methods have been used where the emissions from manure have been allocated to either crop production or livestock production. Usually, the emissions from the use of manure are allocated to crop production (e.g. van Zeijts et al., 1999). However, Dalgaard and Halberg (2007) suggested that the environmental burden of using manure should be considering as a co-product from livestock production. This means that the livestock production system 'pays' all environmental costs related to emissions from manure. However, the livestock system also gets credit for the fertilizer value of the manure. This is reflected in a new guideline from EU on methods for calculating the life cycle environmental performance of products (EU, 2013), which suggests that when manure nitrogen is applied to agricultural land and directly substitutes an equivalent amount of the specific fertiliser nitrogen that the farmer would otherwise have applied, the animal husbandry system from which the manure is derived should be credited for the displaced fertiliser production (taking into account differences in transportation, handling, and emissions) (EU, 2013). This method allows a comparison of the carbon footprint (CF) for different fodder crops on an equal basis.

In this paper we propose a method for calculating the CF of cattle feeds that includes the GHG contribution from changes in soil carbon, and we illustrate how to incorporate into the method the use of cattle manure for crop production. The suggested method is exemplified for typical fodder crops used in Danish dairy production systems.

## 2. Material and methods

An attributional LCA approach was used for calculating the CF of cattle feeds. For each fodder crop, an individual 'food production' system was defined. All crops were fertilized exclusively by artificial fertilizer based on the assumption that the environmental burden of using manure should be considered a co-product from the livestock production. Consequently, 'a manure handling' process was defined to calculate the CF of manure. In addition, a new method for including the GHG contribution from changes in soil carbon was developed based on amounts of C input to soil combined with a newly published model (Petersen et al., 2013) for calculating the proportion of C input that will remain in the soil in a 100-year perspective.

### 2.1. Goal and scope of the study

This paper aims at calculating the CF of the cattle feeds typically used in Danish dairy production systems. It illustrates how to integrate the use of cattle manure for crop production into the method by regarding manure production as a co-product from the dairy system.

The main system studied was the production of fodder crops at dairy farms in Denmark. The contribution from growing, processing and transport was included, as were contributions from changes in soil carbon and from land use change. Fig. 1 shows the system boundary of the feed production system. The functional unit (FU) used for the cattle feed production was '1 kg dry matter (DM) of feed ready to feed'.

Fig. 1 also outlines the system boundary of a dairy system where the cattle feed produced is shown as an important input or a subsystem. The main products are milk and meat with animal manure as an important by-product. A sub-system for handling emissions related to manure production and use was set up. The unit used for the manure production process was '100 kg N ex-animal'.

Finally, the paper explains how total GHG emissions from a dairy system can be calculated by combining the crop production subsystems and the manure handling subsystem with methane emission from enteric fermentation and manure handling as illustrated in the dairy system in Fig. 1.

### 2.2. Calculation of emissions

Factors used for calculating nitrous oxide ($N_2O$) emissions followed IPCC (2006), and for $NH_3$ emission the Danish national norms were used (Mikkelsen et al., 2006; Gyldenkærne and Albrektsen, 2008) (Table 1). Leaching ($NO_3$–N) was calculated as the residual from the surplus of the partial field nitrogen (N) balance (Nielsen and Kristensen, 2005) for growing each feedstuff when all other losses had been deducted, also taking into account changes in soil N. Changes in soil N were assumed to follow the changes in soil C in the proportion 1:10 (Sundberg et al., 1999).

The type of nitrogen fertilizer was assumed to be calcium ammonium nitrate (CAN). The inventory of emissions from the production of CAN fertilizer followed the Danish national average mix of CAN fertilizer (Elsgaard, 2010), with 60% from YARA, produced by a state-of-the-art technique and with reduced $N_2O$ emissions (3.5 kg $CO_2$-eq/kg N) (YARA, 2010) and 40% fertilizer imported from the Baltic countries (5.4 kg $CO_2$-eq/kg N, assuming a 20% improvement in CF compared with Jenssen and Kongshaug (2003)).

### 2.3. LCA Inventory data – crop production

Estimates of annually resource use and output relations for the growing of 1 ha of different crops are presented in Tables 2 and 3. The crops included were winter wheat, spring barley, winter rape, maize, grass clover mixture (60:40), grass and fodder beet. Some of the crops had different end uses: barley was used both for cereals and for silage, grass and grass clover were used for silage, grazing or for grass pellets.

#### 2.3.1. Crop yield and fertilization, crop residues

Crop yield and input of fertilizers for the different crops were based on the Danish national norms (Anonymous, 2010) taking into account that 83% of Danish dairy farms are located on sandy soil and 17% on clay soil (Halberg and Nielsen, 2004). These crop yields were net yields, i.e. the amount that could be fed to the animals. Total yield, i.e. amount harvested or amount that could be harvested, was defined as net yield plus losses in the chain from field to

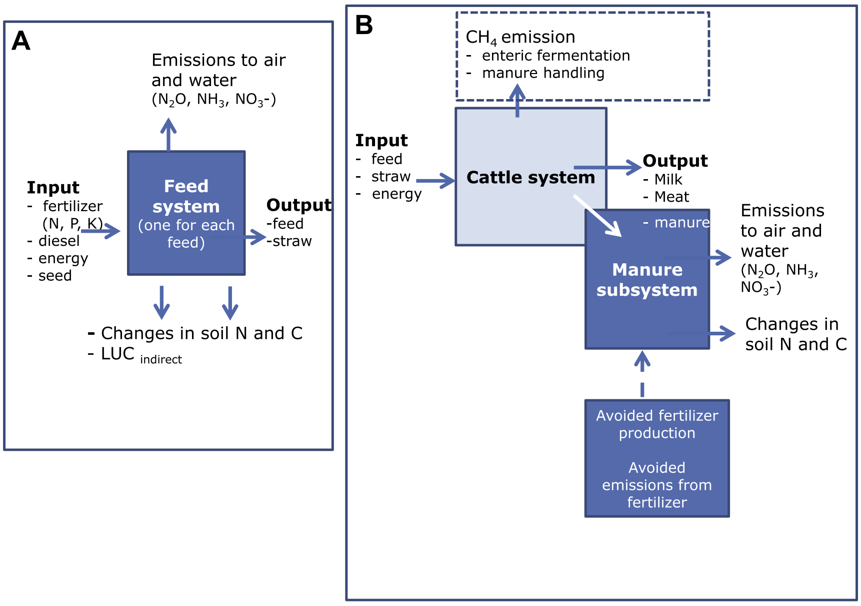


**Fig. 1.** System boundaries of A: A feed production system (one system is set up per feed item) and B: A dairy system, where the manure is a subsystem. The remaining part of the dairy system, $CH_4$ emission (— in broken line) is outside this study.

feed. Some of these losses occur in the field and contribute to the above-ground (AG) crop residues. Tables 2 and 3 show net yield, total yield, total losses and the proportion of losses left in the field, i.e. losses that contribute to AG crop residues. According to Djurhuus and Hansen (2003), 1% of the cereal harvested is lost in the field at harvest. For maize and cereal silage production, a total loss of 13% was assumed, based on Kristensen and Hermansen (1986a), of which the 7% occurred in the field (mainly mechanical losses at harvest), with remaining losses being mainly due to fermentation losses during storage (Kristensen and Hermansen, 1986a). In grass for silage, a total loss of 20% was assumed, based on Kristensen et al. (2006). Here the 7% occurred in the field (mainly mechanical losses at harvest), 6% was mainly due to fermentation losses during storage (Kristensen and Hermansen, 1986a), and the remaining 7% was assumed lost in the fields due to traffic with machinery (Kristensen et al., 2006). For fodder beet, Kristensen et al. (1985) found 19% losses, hereof 10% assumed related to fermentation losses during storage (based on Kristensen and Hermansen, 1986b) and the 9% lost in the field at harvest. In grazed grass clover 40% of the potential yield was assumed left in the field, based on Spörndly and Kumm (2010).

In addition to the above-mentioned losses, stubble and chaff, senescent leaves and beet tops left in the field also contributed to the above-ground (AG) crop residues, and roots left in the soil made up below ground (BG) crop residues. Tables 4 and 5 show the quantities of AG and BG crop residues for different crops and the protein contents of these residues based on typical numbers found for Danish conditions (Djurhuus and Hansen, 2003; Mikkelsen et al., 2005).

BG residues were measured by root wash in winter wheat (three observations), spring barley (17 observations) and winter rape seed (one observation) (Djurhuus and Hansen, 2003). For grass and grass clover in rotation used for silage, 10 root washes were conducted and Djurhuus and Hansen (2003) assumed that the numbers for grass and grass clover for grazing was 10% lower due to the effect of grazing. The DM content of roots in fodder beets was assumed to be 2/3 of that in spring barley based on differences in root structure, whereas maize root amounts were assumed to be the same as for spring barley (Djurhuus and Hansen, 2003).

The amount of stubble was measured in winter rapeseed and spring barley, and the value for spring barley used for winter wheat and maize (Djurhuus and Hansen, 2003). Stubble for grass and grass clover in rotation used for silage was likewise measured (Djurhuus and Hansen, 2003). In the present study, the 'stubble' of grass grazed was assumed to be included in the 40% losses left in the field, whereas Djurhuus and Hansen (2003) assumed that number to be 120% of the stubble from grass for silage based on Whitehead (1995). The amount of leaf senescence in grass for silage was based on Whitehead (1995) and for grass grazed it was assumed to be included in the 40% losses. The amount of chaff was measured for both winter wheat and spring barley (Djurhuus and Hansen, 2003).

### *2.3.2. Energy use*

The amount of diesel used for field operations was based on the number and types of operations set out in farming guidelines for the cultivation of specific crops (Anonymous, 2011a) and the amount of diesel for each operation was based on Dalgaard et al. (2002). It was assumed that the average distance from field to farm was 3 km for both roughage and cereals. Diesel used for transport to the farm was included in the energy used for field work given in Tables 2 and 3. Indirect emissions from machine use were not included in the present study. For 20% of the Danish cattle farms on sandy soil it is possible to irrigate (Kristensen, 2004). The amount of water used per ha for irrigation on sandy soil was based on farming guidelines (Anonymous, 2011a). The electricity use for irrigation was 0.5 kWh per $m^3$ water pumped. The amount of energy used for irrigation in Tables 2 and 3 is an expected average over years and across all farms. In Denmark it is almost always necessary to dry cereals after harvest (Elmholt and Nielsen, 2002). Kristensen and Gundtoft (2003) found an average energy use of 6.8 MJ electricity and 6.2 MJ heat (oil) per 100 kg cereal in a drying plant. The GHG contribution from the production of pesticides and lime was not included in the present study.

## *2.4. Processing and co-product handling*

The concentrated feedstuffs – wheat, barley, and rapeseed cake – are all the result of growing a crop with more than one product.

**Table 1**
Factors for estimation of emissions from crop production, and inventory of input factors.

| | Pollutant | Amount | Emission factor (EF) | Reference EF |
|---|---|---|---|---|
| $N_2O_{direct}$, kg | Housing | Kg N in manure ex animal | | a |
| | - Slurry | | 0.002 | |
| | - Deep litter | | 0.01 | |
| | Storage | Kg N in manure ex housing | | |
| | - Slurry | | 0.005 | |
| | - Deep litter | | 0.005 | |
| | Application | Kg N in manure ex storage | | |
| | - Slurry | | 0.01 | |
| | - Deep litter | | 0.01 | |
| | - At pasture during grazing | | 0.02 | |
| | - Fertilizer | | 0.01 | |
| | Crop residues | kg N pr ha pr year | 0.01 | a |
| $NH_3$–N, kg | Housing | Kg N in manure ex animal | | b |
| | - Slurry | | 0.08 | |
| | - Deep litter | | 0.15 | |
| | Storage | Kg N in manure ex housing | | |
| | - Slurry | | 0.022 | |
| | - Deep litter | | 0.25 | |
| | Application | Kg N in manure ex storage | | |
| | - Slurry | | 0.12 | |
| | - Deep litter | | 0.06 | |
| | - At pasture during grazing | | 0.07 | |
| | - Fertilizer | | 0.022 | |
| | Crop residues | Grass | 0.5 kg/ha | c |
| | | Other arable crops | 2.0 kg/ha | |
| $N_2O$, $_{Indirect}$ kg | From $NH_3$ | $NH_3$–N | 0.01 | a |
| | From leaching | $NO_3$-N[h] | 0.0075 | a |
| **Dairy production system:** | | | | |
| $CH_{4\ enteric}$ | $CH_4$ (MJ/d) = 2.87 + 1.23*DMI-0.1164 *FA[i] | | | d |
| $CH_{4\ manure}$ | Kg $CH_4$= (Feed organic matter + bedding organic matter)* 0.67* $B_o$* MCF[j] | | | a + b |
| | | | **CF, kg $CO_2$-e** | |
| Input | N in fertilizer (per kg N) | | 4.25 | e |
| | P in fertilizer (per kg P) | | 4.63 | f |
| | K in fertilizer (per kg K) | | 0.596 | f |
| | Diesel (per l) | | 3.309 | g |
| | Electricity (from gas, per kWh) | | 0.655 | g |

[a] IPCC, 2006.
[b] Mikkelsen et al., 2006.
[c] Gyldenkærne and Albrektsen, 2008.
[d] Nielsen et al., 2013.
[e] Elsgaard, 2010.
[f] EcoInvent, 2010.
[g] Nielsen et al., 2003.
[h] $NO_3$–N = (Surplus of N balance − other N losses).
[i] DMI = Dry matter intake, kg DM/day, FA = fatty acids, g/kgDM.
[j] Bo = 0.24[a], MCF = 1% though 10% for slurry[b].

**Table 2**
Annual resource use and output from growing crop on 1 ha land.

| Feed | Wheat | Barley | Rape |
|---|---|---|---|
| Input | | | |
| Mineral fertilizer, kg N/ha | 157 | 114 | 181 |
| Mineral fertilizer, kg P/ha | 2.4 | 23 | 32 |
| Mineral fertilizer. kg K/ha | 84 | 49 | 82 |
| Seed, kg | 150 | 150 | 4 |
| Lubricant oil, l | 14 | 11 | 13 |
| Electricity for irrigation, kWh | 105 | 75 | 90 |
| Energy for drying, electricity, kWh | 141 | 92 | 65 |
| Energy for drying, oil, l | 10.9 | 7.1 | 5.1 |
| Field work, MJ[a] | 3784 | 3079 | 3599 |
| Outputn | | | |
| Net cereal yield, kg DM/ha[b] | 6290 | 4110 | 3170 |
| Protein in DM, % | 11.5 | 10.8 | 19.4 |
| Losses left in field, % of DM | 1 | 1 | 1 |
| Total cereal yield, kg DM/ha[c] | 6354 | 4152 | 3202 |
| Total straw yield, kg DM/ha[d] | 3460 | 2267 | 2624 |

[a] 1 L diesel = 37 MJ.
[b] Net yield is the amount that can be fed to animals after reduction for losses.
[c] Total yield is the gross amount before reduction for losses.
[d] In wheat and barley, straw yield is 55% of cereal yield; in rape straw yield is 90% of seed yield (Anonymous, 1996).

Growing wheat and barley results in both straw and grain, and growing rapeseed gives oil, rapeseed cake and straw, which was assumed left in the field as part of the crop residues. In the LCA calculations, economic allocation was used to split the inventory data between the product and co-products.

Rapeseed yield was assumed to be 3430 kg/ha (Table 2), of which 36% is extracted as rapeseed oil and 62% as rapeseed cake (Dalgaard et al., 2008). Total emissions were allocated with 24% to rapeseed cake and 76% to oil yield based on prices from FAO (1998–2008). The energy use for processing 1000 kg rapeseed was assumed to be 50 kWh electricity and 340 MJ heat from oil (Dalgaard et al., 2008). Barley yielded 4840 kg grain and 2667 kg straw. Total emissions were allocated with 95% to grain and 5% to straw based on an average price of barley of 1.18 DKK/kg (2009–2012) (Anonymous, 2012) and 0.12 DKK/kg for straw used as feed/bedding based on the value of the N–P–K in straw (Anonymous, 2011a). Wheat yielded 7400 kg grain and 4070 kg straw. Total emissions were allocated with 95% to grain and 5% to straw based on an average price on wheat of 1.20 DKK/kg (2009–2012) (Anonymous, 2012) and a similar price for straw as for barley. Energy use for processing of 1 tonne of cereals was calculated as the average energy use for rolling (8 kWh/t cereals) and grinding (19 kWh/t) (Mortensen et al., 1981).

For producing 1 tonne of grass pellets (920 kg DM) an input of 1000 kg DM grass clover (from the 'grass clover silage' process) is needed and processing uses 150 kWh electricity, 5040 MJ heat from coal, and 6.8 l diesel (Anonymous, 2011b). Diesel for transport is separately treated below.

### 2.5. Transport

The inventory data for emissions related to transport were taken from the LCAFood database (Nielsen et al., 2003), which is based on data from ETH (Spielmann and Scholz, 2004). Distances and forms of transportation were obtained from the feed industry and the literature. All the roughage was grown on the cattle farm, and transport within the farm was included in the use of diesel. Wheat and barley were transported 25 km from its place of production to the feed factory (16 t lorry) and 25 km from the feed factory to the receiving farm (28 t lorry). Rapeseed was transported 328 km from the place of production to the feed factory (28 t lorry) and rapeseed

**Table 3**
Annual resource use and output from growing crop on 1 ha land.

| Feed | Maize silage | Barley silage | Grass-clover silage | Grass silage | Grass-clover grazed | Grass grazed | Fodder beet |
|---|---|---|---|---|---|---|---|
| Input | | | | | | | |
| Mineral fertilizer, kg N/ha[a] | 151 | 116 | 221 | 346 | 221 | 346 | 168 |
| Mineral fertilizer, kg P/ha | 45 | 30 | 36 | 39 | 36 | 39 | 39 |
| Mineral fertilizer, kg K/ha | 139 | 158 | 211 | 240 | 211 | 240 | 273 |
| Seed, kg | 5 | 150 | 13 | 13 | 13 | 13 | 5 |
| Lubricant oil, l | 18 | 15 | 11 | 11 | 1 | 1 | 13 |
| Electricity for irrigation, kWh | 70 | 100 | 150 | 150 | 160 | 160 | 70 |
| Field work, MJ[b] | 4810 | 3959 | 2923 | 3071 | 222 | 222 | 3515 |
| Output | | | | | | | |
| Net crop yield, kg DM/ha[c] | 11,150 | 7424 | 8272 | 8975 | 7070 | 7701 | 11,494 |
| Protein in DM, % | 7.9 | 10.0 | 17.9 | 17.3 | 24.0 | 22.0 | 7.4 |
| Total losses, % of DM | 13 | 13 | 20 | 20 | 40 | 40 | 19 |
| -% of total losses left in field[d] | 7 | 7 | 14 | 14 | 40 | 40 | 9 |
| Total yield, kg DM/ha[e] | 12,816 | 8533 | 10,340 | 11,219 | 11,783 | 12,835 | 17,790 |

[a] N norm for grass-clover fields: The norm for the crop (234 kg N/ha) + norm for the crop undersown (only once in 2.5 years)(+21 kg N/ha) − reduced N quota in the crop following the grass-clover (−34 kg N/ha) = 221 kg N/ha N norm for grass fields: The norm for the crop (321 kg N/ha) + norm for the crop undersown (once in 2.5 years)(+32 kg N/ha) − reduced N quota in the crop following the grass (−7 kg N/ha) = 346 kg N/ha.
[b] 1 L diesel = 37 MJ.
[c] Net yield is amount fed to cattle.
[d] For fodder beet, besides the 19% loss of total beet yield (2696 kg DM), 3600 kg DM beet top is left in field (Kristensen and Hermansen, 1986b).
[e] Total yield before losses.

**Table 4**
Annual crop residues from growing 1 ha, dry matter (DM), nitrogen (N), carbon (C), and C sequestration per ha.

| Crop | Wheat | | Barley | | Rape | |
|---|---|---|---|---|---|---|
| Straw removed, % | 0[a] Reference crop C input | 100 | 0 | 100 | 0 | 100 |
| Crop residues, kg DM | | | | | | |
| Total above-ground (AG) | 5564 | 2104 | 3695 | 1432 | 3406 | 782 |
| - Losses left in field[b] | 64 | 64 | 42 | 42 | 32 | 32 |
| - Straw left in field[b] | 3460 | 0 | 2263 | 0 | 2624 | 0 |
| - Stubble[c] | 870 | 870 | 870 | 870 | 750 | 750 |
| - Chaff[c] | 1170 | 1170 | 520 | 520 | 0 | 0 |
| Below-ground (BG)[c] | 3450 | 3450 | 1650 | 1650 | 4030 | 4030 |
| Crop residues, kg N | | | | | | |
| Total above-ground (AG) | 36.3 | 18.0 | 25.5 | 11.0 | 22.1 | 5.3 |
| - Losses left in field | 1.0 | 1.0 | 0.7 | 0.7 | 0.9 | 0.9 |
| - Straw left in field | 18.3 | 0 | 14.5 | 0 | 16.8 | 0 |
| - Stubble[c] | 6.3 | 6.3 | 6.3 | 6.3 | 4.4 | 4.4 |
| - Chaff[c] | 10.7 | 10.7 | 4.0 | 4.0 | 0 | 0 |
| Below-ground (BG)[c] | 51.7 | 51.7 | 26.0 | 26.0 | 64.7 | 64.7 |
| N input$_{crop\ residues}$, kg N/ha | 88.0 | 69.7 | 51.5 | 37.0 | 86.8 | 70.0 |
| C input to soil, kg C | | | | | | |
| -from AG crop res.[d] | 2503 | 947 | 1662 | 644 | 1533 | 352 |
| -from BG crop res.[d] | 1553 | 1553 | 743 | 743 | 1814 | 1814 |
| Total C input to soil | 4056 | 2499 | 2405 | 1387 | 3347 | 2165 |
| - C input, corrected for tillage[e] | 4056 | 2499 | 2405 | 1387 | 3347 | 2165 |
| **C input compared to reference crop**[f] | 0 | −1557 | −1651 | −2669 | −709 | −1891 |
| C sequestration[g] | | | | | | |
| - kg C | 0 | −156 | −166 | −267 | −71 | −189 |
| - kg $CO_2$/ha/year[h] | 0 | 571 | 608 | 977 | 260 | 692 |
| **S1: With manure input** | | | | | | |
| **C sequestration**, kg $CO_2$/ha/year[i] | −652 | −81 | −28 | 341 | −391 | 41 |

[a] Wheat with no straw removed and no manure input was used as a reference crop for C input (this type of wheat was assumed to be in C balance, i.e. a soil C change of 0 kg $CO_2$/ha/year).
[b] From Table 2.
[c] Djurhuus and Hansen (2003).
[d] 45% of DM input.
[e] Effect of tillage (IPCC, 2006).
[f] C input of each crop was compared to that of the reference crop 'wheat' by deduction of 4056 kg C/ha/year.
[g] Soil carbon that remains in soil in a 100-year perspective is 10% of input (Petersen et al., 2013). A negative number means C release from soil.
[h] From C to $CO_2$ multiply by 44/12 and change the sign. A negative number now means C sequestration, a positive number the release of carbon from soil.
[i] A scenario was set up with unchanged crop production, except that it included an input of manure corresponding to maximum allowed level according to regulations (Anonymous, 2010): 170 kg total N/ha in most crops, for barley 163 kg total N/ha.

**Table 5**
Annual crop residues from growing 1 ha roughage, dry matter (DM), nitrogen (N), carbon (C), and C sequestration per ha.

| Crop | Maize silage | Barley silage | Grass clover silage | Grass silage | Grass-clover grazed | Grass grazed | Fodder beet |
|---|---|---|---|---|---|---|---|
| Crop residues, kg DM | | | | | | | |
| Total above-ground (AG) | 1767 | 1467 | 4118 | 4241 | 4713 | 5134 | 4877 |
| - Losses left in field[a+b] | 897 | 597 | 1448 | 1571 | 4713 | 5134 | 1277 |
| - Stubble[c] | 870 | 870 | 1670 | 1670 | 0 | 0 | 0 |
| - Leaf senescence[c] | 0 | 0 | 1000 | 1000 | 0 | 0 | 0 |
| - Beet top | 0 | 0 | 0 | 0 | 0 | 0 | 3600 |
| Below-ground (BG)[c] | 1650 | 1650 | 3180 | 3180 | 2860 | 2860 | 1100 |
| Crop residues, kg N | | | | | | | |
| Total above-ground (AG) | 17.7 | 15.9 | 83.7 | 85.8 | 181 | 181 | 109.6 |
| - Losses left in field[b] | 11.4 | 9.5 | 41.4 | 43.5 | 181 | 181 | 15.1 |
| - Stubble[c] | 6.3 | 6.3 | 32.3 | 32.3 | 0 | 0 | 0 |
| - Leaf senescence[c] | 0 | 0 | 10 | 10 | 0 | 0 | 0 |
| - Tops | 0 | 0 | 0 | 0 | 0 | 0 | 94.5 |
| Below-ground (BG)[c] | 26.0 | 26.0 | 46.8 | 46.8 | 42.1 | 42.1 | 17.3 |
| **N input crop residues, kg N**[d] | 43.7 | 41.9 | 65.3 | 66.3 | 111.6 | 111.6 | 126.9 |
| C input to soil, kg C | | | | | | | |
| - from AG crop residues.[e] | 795 | 660 | 1853 | 1908 | 2121 | 2310 | 2195 |
| - from BG crop residues.[e] | 743 | 743 | 1431 | 1431 | 1287 | 1287 | 495 |
| Total C input to soil | 1538 | 1403 | 3284 | 3339 | 3408 | 3597 | 2690 |
| - C input, corrected for tillage[f] | 1538 | 1403 | 3777 | 3840 | 3919 | 4137 | 2690 |
| **C input compared to reference crop**[g] | −2518 | −2653 | −279 | −216 | −137 | 81 | −1366 |
| C sequestration, | | | | | | | |
| - kg C[h] | −252 | −265 | −28 | −22 | −14 | 8 | −137 |
| - kg $CO_2$/ha/year[i] | 922 | 974 | 106 | 81 | 51 | −29 | 501 |
| **S1: With manure input**[j] **C sequestration,** kg $CO_2$/ha/year | 271 | 338 | −545 | −570 | −692 | −793 | −150 |

[a] Contribution from leaves decay is included in losses left in field for grass and grass clover grazed.
[b] From Table 3.
[c] Djurhuus and Hansen (2003).
[d] For grass and grass clover, fraction revenue was taken into account with a factor 0.5 due to revenue every second year (as suggested by IPCC, 2006 11.7A).
[e] 45% of DM input.
[f] Effect of tillage as suggested by IPCC (2006).
[g] C input compared to that of the reference crop 'wheat' by deduction of 4056 kg C/ha/year.
[h] Soil carbon that remains in soil in a 100 year perspective is 10% of input according to Petersen et al. (2013). Here a positive number means an input of carbon to soil.
[i] From C to $CO_2$ multiply by 44/12 and change the sign. A negative number now means C sequestration, a positive number the release of carbon from soil.
[j] A scenario was set up with input of manure corresponding to maximum allowed level according to regulations (Anonymous, 2010): 170 kg total N/ha in most crops, for barley 163 kg total N/h and for grazed grass and grass-clover the amount of manure deposited on that field corresponded to that from grazing cattle that were 100% grass fed, here input of 223 and 212 kg total N in grass-clover and grazed grass, respectively.

cake 168 km from the feed factory to the receiving farm (28 t lorry). Grass was transported 21 km from the place of production to the feed factory (28 t lorry) and the grass pellets 134 km from the feed factory to the receiving farm (28 t lorry).

### 2.6. Soil carbon (C) sequestration

The GHG contribution from soil C changes caused by the crop production was estimated in accordance with a new approach suggested by Petersen et al. (2013). The use of this approach has been illustrated by Knudsen et al. (2014). The approach by Petersen et al. (2013) is based on a single year's addition of C (from crop residues, etc.) and the associated effect on atmospheric $CO_2$. Petersen et al. (2013) estimated that 10% of the C added to the soil will be sequestered in a 100-year perspective. The input of carbon to soil was based on the input of crop residues, the sum of above- (AG) and below-ground (BG) with an assumed C content of 45% of dry matter.

The approach by Petersen et al. (2013) is valid for showing differences in C sequestration over time or between crops. Wheat is a common crop in the European landscape and the amount of C input to the soil from wheat crop residues etc. determines the equilibrium of carbon in the soil under wheat cultivation. Furthermore, based on Danish measurements of soil C changes (Heidmann et al., 2001), wheat grown without manure input and with no straw removed was assumed to have a C sequestration close to 0 g C/ha/year. Thus, assuming that the amount of C input from wheat cultivation represents the average C input for European soils, crops with a lower C input than 'wheat with no straw removed' would result in a carbon loss from soil to the atmosphere over time and crops with a higher C input would result in carbon sequestration over time compared to the present average soil equilibrium. Therefore, 'wheat grown without manure input and with no straw removed' was chosen as a reference crop. The difference in total C input from the wheat crop was calculated for each crop and multiplied by 10% (Petersen et al., 2013) to get the effect of soil carbon changes on atmospheric $CO_2$. This model does not include the effect of tillage. Thus, this was added as suggested by IPCC (2006) with a tillage factor of 1.15 for no-till (grass and grass-clover) and 1.00 for full tillage (all other crops).

All crops in Tables 4 and 5 were grown without input of manure. To illustrate the effect of manure input, a scenario was set up for each crop where some fertilizer was replaced with input of manure corresponding to the maximum allowed level of manure in Danish regulations (Anonymous, 2010), which is 170 kg manure N/ha for most crops, but 163 kg for barley.

### 2.7. Land use change

LUC was calculated according to Audsley et al. (2009) where all use of land for crop production is assumed to increase the pressure on land use and thus causing LUC somewhere in the world. LUC

**Table 6**
Annual N and P budgets at field level from growing 1 ha.

| | N balance, kg/ha | | | P balance, kg/ha | | |
|---|---|---|---|---|---|---|
| Crop | Wheat | Barley | Rape | Wheat | Barley | Rape |
| Straw removed, % | 100 | 100 | 0 | 100 | 100 | 0 |
| Input | | | | | | |
| Mineral fertilizer | 157 | 114 | 181 | 24 | 23 | 32 |
| Seed | 2 | 2 | 0 | 0.5 | 0.4 | 0 |
| Fixation | 0 | 0 | 0 | – | – | – |
| Deposition | 15 | 15 | 15 | – | – | – |
| Total input | 174 | 131 | 196 | 24 | 23 | 32 |
| Output | | | | | | |
| Net crop yield | 116 | 71 | 98 | 22 | 16 | 24 |
| Straw | 18 | 15 | 0 | 3 | 2 | 0 |
| Total output | 134 | 86 | 98 | 25 | 18 | 24 |
| **Field balance** | 40 | 46 | 98 | 0 | 6 | 8 |
| Losses | | | | | | |
| $NH_3$–N | 5.5 | 4.5 | 6.0 | | | |
| NO–N[a] | 0.8 | 0.6 | 0.8 | | | |
| $N_2O$–N, direct | 2.4 | 1.6 | 2.7 | | | |
| $N_2$–N[b] | 5.8 | 3.8 | 6.6 | | | |
| $N_2O$–N, indirect | 0.4 | 0.5 | 0.7 | | | |
| Soil change, N or P[c] | −15.6 | −26.7 | −7.1 | 0 | 5.6 | 7.7 |
| Difference (potential leaching) kg $NO^3$–N or $PO_4^{3-}$–P[d] | 40.7 | 61.4 | 88.3 | 0 | 0.2 | 0.2 |

[a] NOx-N (=NO + $NO_2$, where $NO_2$ is assumed to be negligible) is calculated based on the NOx-N : $NH_3$–N ration of 12:88 (Schmidt and Dalgaard, 2012) and the known amount of $NH_3$–N.
[b] Total denitrification ($N_2$ + $N_2O$) was calculated by SimDen (Vinter and Hansen, 2004).
[c] Changes in soil N: Based on C:N ratio of 10:1 (Sundberg et al., 1999) and C added to soil from Table 4 ('C sequestration, kg C'). Positive value means buildup in soil. Changes in soil P: 97% of surplus will remain in soil, 3% lost through leaching (Dalgaard et al., 2006).
[d] Potential leaching is calculated as the difference between field balance and other losses.

causes a release of 8.5 Gt $CO_2$-eq per year, to which agriculture contributes 58%. This gives a contribution of 1.43 t $CO_2$-eq per ha when divided by the total agricultural area of 3475 Mha (Audsley et al., 2009). In the present study, LUC was included by multiplying land use ($m^2$/kg DM feed) by an LUC factor of 143 g $CO_2$-eq/$m^2$.

### 2.8. Manure systems

We use the approach described by Dalgaard and Halberg (2007) to account for the environmental cost of using manure, considering it as a co-product from the livestock production. This means, that the livestock production system 'pays' all environmental costs

**Table 7**
Annual N and P budgets at field level from growing 1 ha.

| | N balance, kg/ha | | | | | | | P balance, kg/ha | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Feed | Maize silage | Barley silage | Grass-clover[a] silage | Grass silage | Grass- clover[a] grazed | Grass grazed | Fodder beet | Maize silage | Barley silage | Grass – clover silage | Grass silage | Grass- clover grazed | Grass grazed | Fodder beet |
| Input | | | | | | | | | | | | | | |
| Mineral fertilizer | 151 | 116 | 221 | 346 | 221 | 346 | 168 | 45 | 30 | 36 | 39 | 36 | 39 | 39 |
| Seed, kg | 0.2 | 2.2 | 0.5 | 0.5 | 0.5 | 0.5 | 0.2 | 0.03 | 0.4 | 0.08 | 0.08 | 0.08 | 0.08 | 0.03 |
| Fixation[b] | 0 | 0 | 113 | 0 | 113 | 0 | 0 | – | – | – | – | – | – | – |
| Deposition | 15 | 15 | 15 | 15 | 15 | 15 | 15 | – | – | – | – | – | – | – |
| Total input | 166 | 133 | 349 | 362 | 350 | 362 | 183 | 45 | 30 | 36 | 39 | 36 | 39 | 39 |
| Output | | | | | | | | | | | | | | |
| Net crop yield | 141 | 119 | 237 | 248 | 272 | 271 | 136 | 26 | 21 | 33 | 36 | 28 | 26 | 20 |
| **Field balance** | 25 | 14 | 112 | 113 | 78 | 91 | 47 | 19 | 10 | 3 | 3 | 8 | 13 | 20 |
| Losses | | | | | | | | | | | | | | |
| $NH_3$–N | 5.3 | 4.6 | 5.4 | 8.1 | 5.4 | 8.1 | 5.7 | | | | | | | |
| NO-N[c] | 0.7 | 0.6 | 0.7 | 1.1 | 0.7 | 1.1 | 0.8 | | | | | | | |
| $N_2O$–N, direct | 1.9 | 1.6 | 2.9 | 4.1 | 3.3 | 4.6 | 2.9 | | | | | | | |
| $N_2$–N[d] | 4.8 | 4.1 | 7.6 | 10.2 | 8.0 | 11.2 | 5.4 | | | | | | | |
| $N_2O$–N, indirect | 0.3 | 0.3 | 0.8 | 0.8 | 0.5 | 0.6 | 0.4 | | | | | | | |
| Soil change, N or P[e] | −25.2 | −26.6 | −2.9 | −2.2 | −1.4 | 0.8 | −13.7 | 18.8 | 9.4 | 2.9 | 3.1 | 7.6 | 12.5 | 18.9 |
| Difference (potential leaching) kg $NO^3$–N or $PO_4^{3-}$-P[f] | 37.1 | 29.4 | 97.5 | 90.9 | 61.4 | 64.6 | 45.6 | 0.6 | 0.3 | 0.1 | 0.1 | 0.2 | 0.4 | 0.6 |

[a] 60% grass and 40% clover.
[b] Fixation in grass-clover: 100 kg N/ha + 12.5 kg N/ha from undersown grass-clover (Kristensen. I.S., 2013, pers. comm.).
[c] NOx-N (= NO + $NO_2$, where $NO_2$ is assumed to be negligible) is calculated based on the NOx-N : $NH_3$–N ration of 12:88 (Schmidt and Dalgaard, 2012).
[d] Total denitrification ($N_2$ + $N_2O$) was calculated by SimDen (Vinter and Hansen, 2004).
[e] Changes in soil N: Based on C:N ratio of 10:1 and C added to soil from Table 5. Positive value means buildup in soil. Changes in soil P: 97% of surplus will remain in soil, 3% lost through leaching (Dalgaard et al., 2006).
[f] Leaching is calculated as the difference between field balance and other losses.

**Table 8**
Contribution to carbon footprint (CF) of feed ready to feed, g $CO_2$/kg DM and land use.

| Feed | Wheat grain[b] | Wheat straw[b] | Barley Grain[a] | Barley Straw[a] | Rapeseed | Rapeseed cake[c] | Grass pellets |
|---|---|---|---|---|---|---|---|
| CF, g $CO_2$/kg DM | | | | | | | |
| - Growing | 406 | 40 | 484 | 49 | 963 | 390 | 439 |
| - Processing | 11 | 1 | 11 | 1 | 0 | 28 | 715 |
| - Transport | 18 | 18 | 18 | 18 | 122 | 75 | 38 |
| Total CF | 434 | 59 | 512 | 68 | 1085 | 494 | 1190 |
| C sequestration[d] | 86 | 8 | 225 | 22 | 82 | 34 | 14 |
| $LUC_{indirect}$[e] | 215 | 21 | 328 | 33 | 451 | 182 | 188 |
| Total + $C_{seq}$ | 520 | 67 | 736 | 91 | 1168 | 528 | 1204 |
| Total + $C_{seq}$ + $LUC_{indirect}$ | 735 | 88 | 1065 | 124 | 1618 | 710 | 1392 |
| Land use, $m^2$ | 1.51 | 0.15 | 2.31 | 0.24 | 3.16 | 1.28 | 1.32 |

[a] CF for 'barley with 100% straw removed' economic allocation with 95% to barley and 5% to straw.
[b] CF for 'wheat with 100% straw removed', economic allocation with 95% to wheat and 5% to straw.
[c] CF for 'rapeseed', economic allocation with 24% to rapeseed cake. Transport of rape seed to feed factory allocated too.
[d] From Table 3.
[e] 143 g $CO_2/m^2$ used (Audsley et al., 2009).

related to emissions from manure in housing and storage and in the case where emissions from the spreading of manure exceed emissions from spreading the same amount of fertilizer. However, the livestock system is also credited for the fact that the use of manure reduces the amount of artificial fertilizer being used. The saved amount of N fertilizer was calculated as the total N content in the manure after losses multiplied by the percentage of N that is supposed to be available for crops (Anonymous, 2010).

Both cattle feed production and manure production are subsystems of the dairy system, as illustrated in Fig. 1. Three different manure production systems were included in the study: 1) manure deposit on pasture, 2) manure as slurry and 3) manure as deep litter. Input to each manure system is N ex animal, calculated according to Kristensen and Kristensen (2007) as N in feed minus N in grain and milk. Manure as deep litter has an input of straw (see Table 10) whereas no straw is assumed used in manure as slurry.

## 3. Results

### 3.1. Nitrogen and phosphorus balances and distribution of losses

Tables 6 and 7 show the partial N and P budgets for the different crops. For all crops, except grazed grass, there was a release of N from soil, which increased the calculated leaching compared to the calculation in the IPCC guidelines (2006) that includes 30% of N input in fertilizer and manure but excludes the contribution from changes in soil N.

### 3.2. Crop residues, C input to soil and sequestration

Input to soil from crop residues was presented as dry matter (DM), C and N (Tables 4 and 5). The highest input of above-ground residues (AG) was as expected from grazed grass and grass-clover. Wheat with all straw ploughed in and fodder beet also had a high input of AG crop residues of around 5 t DM/ha due to high inputs from straw (3.5 t) and beet top (3.6 t). The lowest input of AG crop residues was found for maize and barley for silage. Wheat also had a surprisingly high input of below-ground (BG) crop residues. The quantity was 2.6 times higher than if calculated from the relation between BG:AG as given by the IPCC (2006). Also in rape was the measured contribution from BG crop residues higher than expected according to IPCC. Barley sown in spring only had half the amount of BG residues compared with wheat sown in autumn. Crop residues from grass and grass-clover to silage and barley for silage were in accordance with IPCC (2006).

For the reference crop of wheat with no straw removed and no input of manure, soil carbon sequestration was an estimated 0 kg $CO_2$/ha. If instead all straw was removed from the wheat field, 571 kg $CO_2$ would be released per ha every year. Growing barley or rapeseed caused C release even with 100% of the straw ploughed in. The highest C release was seen in maize and barley for silage, whereas growing grass oscillated around C balance (from 106 kg $CO_2$ released to 29 kg C sequestrated).

All fodder crops in Tables 4 and 5 were fertilized by artificial fertilizer only and the C sequestration therefore had no C input from manure. The scenarios (S1), on the other hand, assumed an input of the maximum amount of manure to each crop. These scenarios showed that with the use of manure in crop production, all cereals sequestered C if straw was incorporated in the soil and wheat sequestered C even with all straw removed (Table 4). In roughage production, the use of manure resulted in C sequestration for all crops, except for maize and barley for whole crop silage production (Table 5). The highest levels of C sequestration were seen in the grazed crops.

**Table 9**
Contribution to carbon footprint (CF) of feed ready to feed, g $CO_2$-eq/kg DM and land use.

| Feed | Maize silage | Barley silage | Grass-clover silage | Grass silage | Grass-clover grazed | Grass grazed | Fodder beet |
|---|---|---|---|---|---|---|---|
| CF, g $CO_2$-eq/kg DM | | | | | | | |
| - Growing | 224 | 285 | 404 | 503 | 448 | 565 | 264 |
| - Processing | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| - Transport | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Total CF | 224 | 285 | 404 | 503 | 448 | 565 | 264 |
| C sequestration[a] | 83 | 131 | 13 | 9 | 7 | −4 | 44 |
| $LUC_{indirect}$[b] | 128 | 193 | 173 | 159 | 202 | 186 | 124 |
| Total CF + $C_{seq}$ | 307 | 416 | 417 | 512 | 455 | 561 | 308 |
| Total CF + $C_{seq}$ + $LUC_{indirect}$ | 435 | 609 | 590 | 671 | 657 | 747 | 432 |
| Land use, $m^2$ | 0.90 | 1.35 | 1.21 | 1.11 | 1.41 | 1.30 | 0.87 |

[a] From Table 4.
[b] 143 g $CO_2/m^2$ used (Audsley et al., 2009).

**Table 10**
The manure subsystem, greenhouse gas (GHG) emissions from 100 kg total N ex-animal.

| Housing system<br>Manure system | Outdoor<br>At pasture | Indoor<br>Slurry | Indoor[a]<br>Deep litter |
|---|---|---|---|
| **Emissions from manure handling:** | | | |
| **$N_2O$–N direct, kg** | | | |
| -housing | 0 | 0.2 | 1.0 |
| -storage | 0 | 0.5 | 0.5 |
| -application | 2.0 | 1.0 | 1.0 |
| **$NH_3$–N, kg** | | | |
| -housing | 0 | 8.0 | 15.0 |
| -storage | 0 | 2.2 | 25.0 |
| -application | 7.0 | 12.0 | 6.0 |
| **$N_2O$–N indirect, kg** | | | |
| -from $NH_3$–N | 0.07 | 0.22 | 0.46 |
| -from leaching [b] | 0.68 | 0.58 | 0.39 |
| (1) **Total GHG from manure handling, kg $CO_2$-eq** | **1288** | **1171** | **1569** |
| **C sequestration from manure** | | | |
| N input to soil after losses, kg N [c] | 90 | 75 | 58 |
| Related C input to soil, kg C [d] | 939 | 783 | 1581 |
| Soil C remaining in soil), kg soil C [e] | 94 | 78 | 158 |
| (2) **Total GHG from C sequestration, kg $CO_2$-eq** [f] | **−344** | **−287** | **−579** |
| **N from manure stored in soil and reduced leaching** [g] | | | |
| N stored in soil, kg N | 9.4 | 7.8 | 15.8 |
| Saved indirect $N_2O$ emissions, kg $N_2O$–N | 0.07 | 0.06 | 0.12 |
| (3) **GHG from avoided leaching, kg $CO_2$-eq** | **−21** | **−17** | **−55** |
| (4) **Total GHG from manure handling, kg $CO_2$-eq** [h] | **923** | **867** | **935** |
| **Avoided fertilizer production:** | | | |
| **Fertilizer value of manure** | | | |
| N, kg [i] | 70 | 70 | 45 |
| P, kg [i] | 14 | 14 | 20 |
| K, kg [i] | 91 | 91 | 137 |
| GHG from avoided fertilizer prod., kg $CO_2$-eq | | | |
| -N [j] | −298 | −298 | −191 |
| -P | −67 | −67 | −93 |
| -K | −54 | −54 | −82 |
| (5) **GHG from avoided fertilizer prod., kg $CO_2$-eq** | **−418** | **−418** | **−366** |
| **Avoided emission from fertilizer** | | | |
| $N_2O$-$N_{direct}$, kg from spreading [j] | 0.7 | 0.7 | 0.45 |
| $NH_3$–N, kg from spreading | 1.54 | 1.54 | 0.99 |
| $N_2O$-$N_{indirect}$, kg from $NH_3$ and leaching | 0.53 | 0.53 | 0.34 |
| (6) **GHG from avoided fertilizer emission, kg $CO_2$-eq** | **−574** | **−574** | **−370** |
| (7) **Total GHG from avoided fertilizer** [k] | **−992** | **−992** | **−736** |
| (8) **GHG from 100 kg N of different types, kg $CO_2$-eq** [l] | **−69** | **−125** | **199** |

13) GHG(8) = GHG(4) + GHG(7).
[a] CF from import of straw is not included in this calculation (see Table 11).
[b] Leaching ($NO_3$–N) calculated as input minus other emission.
[c] Input to soil is 100 kg N ex animal minus all losses.
[d] In the deep litter system there is an extra N input from straw. Per 100 kg N ex animal there is an input of 2581 kg straw: 10 kg straw/cow/day (Anonymous, 2008), 141.4 kg N ex animal/cow/year (Poulsen, 2011). 14 kg N from straw, after losses 7.2 kg N as input to soil from straw.
[e] C:N in manure deposited at pasture and in slurry 8:1 (Wesnæs et al., 2009) and C:N in deep litter of 21:1 (Osada et al., 2001) both multiplied by a factor of 1.3 (Petersen, B pers comm., 2013).
[f] The model by Petersen et al. (2013).
[g] C to $CO_2$ factor multiplication 44/12.
[h] CF and EF from Table 1.
[i] Per 10 kg C stored in soil, 1 kg N is stored in soil (Sundberg et al., 1999).
[j] Anonymous, 2010.
[k] GHG(4) = GHG(1) + GHG(2) + GHG(3).
[l] GHG(7) = GHG(5) + GHG(6).

### 3.3. *Carbon footprint of feed − including LUC and soil C changes*

Tables 8 and 9 present the LCA results on the CF of feed as the sum of contributions from growing, processing, transport, plus the contribution from carbon sequestration and $LUC_{indirect}$. The ranking of feedstuffs did not change much, irrespective of whether the GHG contributions from soil C changes and LUC were included or not. In all cases, straw, maize silage and fodder beet had the smallest CF and rapeseed and grass pellets had the highest CF. With the contribution from soil C changes and LUC included, barley moved up among the feeds with the highest CF. Including LUC and soil C changes in the CF of feed resulted in a large group of feeds having a CF between 600 and 700 g $CO_2$-eq/kg DM, including wheat, rapeseed cake, barley silage and grass for silage or grazed. In Fig. 2, the animal feeds grown in Denmark are ranked according to total CF including the contributions from LUC and soil carbon.

### 3.4. *Carbon footprint of feed combined with effect of using manure, and CF of milk*

The effect of producing and using manure, calculated per 100 kg N ex animal either deposited at pasture or collected as slurry or deep litter, is shown in Table 10. Emissions related to manure handling (housing, storage and application) varied from 1171 kg $CO_2$-eq per 100 kg N ex animal in a slurry system to 1569 kg $CO_2$-eq/100 kg N ex animal in a deep litter system.

Application of manure caused C sequestration in soil, highest for the deep litter system that also included input of C from straw. No straw was assumed used in the two other systems. If carbon was sequestrated in soil also N was accumulated in soil which causes a lower risk for leaching. These effects are also credited. Altogether, the manure system can deduct 304 kg $CO_2$-eq, 365 kg $CO_2$-eq and 634 kg $CO_2$-eq per 100 kg N ex animal in, respectively, slurry, pasture and deep litter systems, due to soil C sequestration reducing emissions and reduced N leaching caused by N accumulation in soil.

As an application of manure substitutes the use of a quantity of artificial fertilizer, the saved emissions from production and spreading of that quantity fertilizer amounted to 992 kg $CO_2$-eq per 100 kg N ex animal in the slurry and pasture system, but only 736 kg $CO_2$-eq per 100 kg N ex animal in the deep litter system. This is because the substitution rate of kg fertilizer N per kg N ex animal in Danish legislation is lower at only 0.45 for deep litter compared with 0.7 for slurry and N deposit at pasture (Anonymous, 2010). However, when looking at the calculated ‘N input to soil after losses’, these numbers were higher for N deposited at pasture (90 kg N/100 kg N ex-animal) and for deep litter (58 kg N/100 kg N ex-animal) than the above-mentioned numbers given by Danish legislation. There is obviously a potential for higher utilization of these types of manure. When totalling the effects of using manure in the three different ways, GHG emissions from the dairy system were reduced by 125 kg $CO_2$-eq per 100 kg N ex animal as slurry and by 69 kg $CO_2$-eq/100 kg N ex animal deposit at pasture, whereas there was a release of 199 kg $CO_2$-eq per 100 kg N ex animal as deep litter.

Table 11 gives a sample illustration of how the CFs of feedstuffs can be combined with the CFs of manure handling. Three different scenarios were set up representing a dairy cow for one year. The CF of the total ration fed to the cows is shown using the CF per feed (Tables 8 and 9). Based on intake of N and production of milk and meat, N ex animal was 169 kg N per year for a cow in the pasture system. In the pasture system, the cow was only on grass during the summer and only for some of the day. The proportion of N excreted at pasture was calculated from the proportion of DM intake from pasture (68%). The GHG emissions from manure handling of −146 kg $CO_2$-eq/cow/year was then arrived at by taking the sum of 169 kg N ex animal*0,68*(-69 kg $CO_2$-eq/100 kg N ex animal deposited at pasture) (Table 10) and 169 kg N ex animal *0,32*(−125 kg $CO_2$-eq/100 kg N ex animal deposited as slurry). GHG emissions from feed production and manure handling were

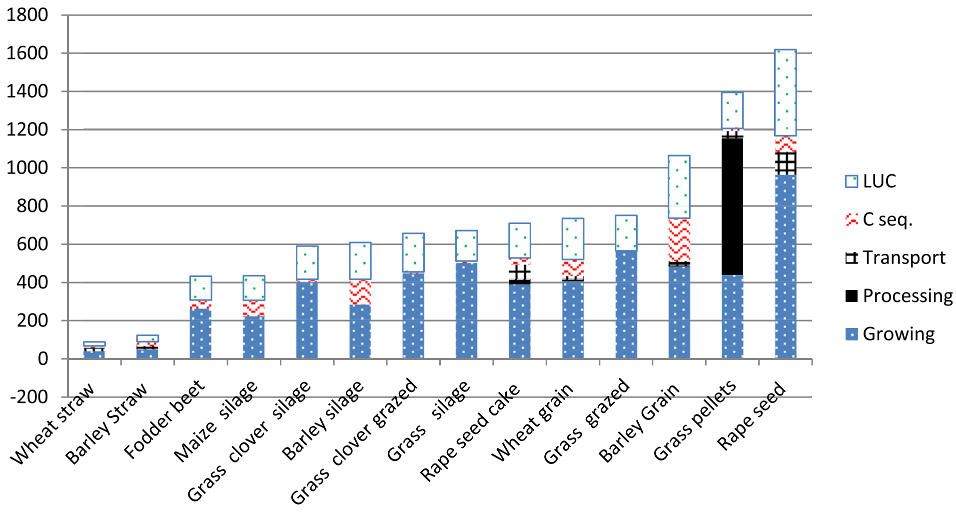


**Fig. 2.** Carbon footprint of animal feeds, g $CO_2$/kg DM.

combined with the emissions from the remaining part of the cattle system – the $CH_4$ emission from enteric fermentation and manure handling (see Fig. 1 and EF in Table 1). Altogether this produces the CF of milk production taking into account soil C and N changes and LUC. This CF was found to vary from 1.08 kg $CO_2$-eq/kg milk in the slurry system to 1.14 g $CO_2$-eq/kg milk in the deep litter system.

## 4. Discussion

The suggested approach of dividing the dairy system into subsystems means that it is possible to evaluate the GHG effect of different methods of producing milk by adding the GHG contributions from the feed production and the manure system using the figures given in this paper plus the contribution from methane emissions by using relevant equations.

Separating systems into subsystems does, of course, have to be treated with care as there might be some relations and interrelations that are not taken properly into account. A critical element of applying the figures for the CF of feed presented in this paper more generally, even in a North European context, is that the crop – although defined as a single crop – is dependent on figures relating to crops grown in a crop rotation system. This is especially critical for the C turnover, but also for the N losses defined for each crop. For example, some of the N accumulated in a grass field will initially be lost when the crop is ploughed in. Furthermore, the N supply, yield levels, etc., in cereals are very dependent on whether the cereal is produced following a grass crop or other crops. However, even with these reservations in mind, the approach is comprehensive, where parts can easily can be adapted to other production systems, other types of crops or other emission factors in manure systems.

### *4.1. Comparison with LCA results from similar studies*

In a Dutch study (FeedPrint, 2012; Vellinga et al., 2013), the CF was estimated for several feeds based on the contribution from growing, processing and transport plus a LU-LUC contribution. Consistent with our findings, they found a small CF contribution from maize silage, a large CF for rapeseed, and a big middle group with feeds like grass pasture or grass for silage and wheat grain for which the total CF ranged from 600 to 700 g $CO_2$-eq/kg DM. However, Vellinga et al. (2013) found straw and fodder beets to rank much higher, due to the higher contribution from the growing stage and a smaller CF from barley due to the smaller contribution from LU-LUC than in our study. As they similar to us based LUC on Audsley et al. (2009), the discrepancy was caused by differences in the contribution from soil C. Vellinga et al. (2013) assumed an emission of soil C per ha per year of 30 kg C for all arable crops, which is much lower than our findings for barley based on C input to soil.

A Swedish study (Flysjö et al., 2008; SIK Foder, 2013) estimated the CF from feed, though without including the contribution from LUC or soil carbon changes. Their results are very consistent with ours. In both studies, the CFs of maize and barley silage are low (below 300 g $CO_2$-eq/kg DM), the CF of rapeseed high (>880 g $CO_2$-eq/kg DM) and grain, grass and rapeseed cake fall in the middle (370–460 g $CO_2$-eq/kg DM). This was found despite the input of manure in the Swedish study being seen as a by-product from livestock production that the crop system gets for free, although 'paying' for the emissions caused by the use of manure.

### *4.2. LUC*

In the present study, LUC was included by multiplying land use ($m^2$/kg DM feed) by an LUC factor of 143 g $CO_2$-eq/$m^2$, as suggested by Audsley et al. (2009). If the contribution from LUC was instead included with a product-based approach (BSI, 2011), none of the crops in the present study would have a contribution from LUC as none were grown in regions where deforestation takes place. If, on the other hand, LUC was included as suggested by Schmidt et al. (2012), who assumed that the marginal effect of including 1 extra ha of land (when also the productivity of the land was taken into account) has a global average of 783 g $CO_2/m^2$, the relative contribution from LUC would increase but there would only be minor changes in the ranking of the CF of feeds.

### *4.3. Soil carbon changes*

Regarding soil C sequestration, both Vleeshouwers and Verhagen (2002) and Vellinga et al. (2004) assumed that growing grass would work as a sink for C, whereas growing other crops would cause a net release of C from soil. However, their estimated, values for carbon sequestration in grass differ a lot, 191 g and 15 g $CO_2/m^2$/year, respectively. In our study we only found soil C sequestration in grass if growing grass was combined with the use of manure. The level of C sequestration in grass in our study ranged

**Table 11**
An example showing greenhouse gas (GHG) emission from one dairy cow for one year in three different housing systems. Contribution from feed production and manure handling was combined with the remaining part of the cattle system, the $CH_4$ emission.

| Housing system | Pasture | Slurry | Deep litter |
|---|---|---|---|
| Manure handling, % of N ex animal[a] | | | |
| - deposit at pasture | 68 | 0 | 0 |
| - slurry | 32 | 100 | 0 |
| - deep litter | 0 | 0 | 100 |
| Feeding, kg DM/cow/year[b] | | | |
| Grazed grass-clover | 4560 | 0 | 0 |
| Grass-clover silage | 0 | 980 | 980 |
| Maize silage | 900 | 3050 | 3050 |
| Straw, feeding | 60 | 90 | 90 |
| Barley | 880 | 640 | 640 |
| Rapeseed cake | 0 | 1520 | 1520 |
| Grass pellets | 410 | 580 | 580 |
| Total feed input, kg DM | 6810 | 6860 | 6860 |
| *Straw for bedding, kg* | *0* | *0* | *3650* |
| GHG from feed production, kg $CO_2$-eq[c] | | | |
| -Growing, processing and transport | 3190 | 3953 | 4164 |
| -Soil C | 1565 | 480 | 549 |
| -LUC indirect | 1405 | 1315 | 1417 |
| **Total GHG from feed, kg $CO_2$-eq** | **6160** | **5748** | **6131** |
| Cow N balance per year | | | |
| N in feed, kg[d] | 217 | 179 | 179 |
| N in milk and gain[e] | 48 | 48 | 48 |
| **N ex animal** | **169** | **131** | **131** |
| GHG from manure handling, kg $CO_2$-eq[f] | | | |
| -Emissions | 2114 | 1534 | 2055 |
| -Saved fertilizer production and emissions | −1676 | −1300 | −964 |
| -Effect on soil C and N of manure | −584 | −398 | −831 |
| **GHG from manure, kg $CO_2$-eq** | **−146** | **−164** | **260** |
| **Total GHG from feed and manure, kg $CO_2$-eq/cow/year (A)** | **6013** | **5584** | **6391** |
| (g $CO_2$-eq/kg milk) | (0.67) | (0.62) | (0.71) |
| The remaining cattle system (B) | | | |
| $CH_4$ from enteric fermentation, kg $CH_4$/cow/year | 154 | 148 | 148 |
| $CH_4$ from manure handling, kg $CH_4$/cow/year | 6,7 | 17,0 | 5,3 |
| GHG from $CH_4$, kg $CO_2$-eq/cow/year | **4018** | **4125** | **3833** |
| **GHG per cow per year (A + B)** | **10031** | **9709** | **10224** |
| **GHG, g $CO_2$-eq/kg milk** | **1.11** | **1.08** | **1.14** |
| Total land use, $m^2$/cow/year[g] | 9825 | 9196 | 9713 |

[a] In the 'pasture' housing system the cows are indoors during milking and some additional feeding. Distribution of manure between at pasture and in housing according to energy from grass relative to energy fed in housing.
[b] Same energy input of feed and milk production for the three systems.
[c] Using the numbers in Tables 9 and 10, including straw for bedding.
[d] Based on standard crude protein content (Møller et al., 2005).
[e] 9000 kg milk with 3.3% protein and 1.7 kg N from gain and fetus.
[f] This GHG takes into account the amount of fertilizer that is substituted by manure, effect on soil C and N etc. GHG was calculated based on amount of 'N ex animal' (this table) and GHG per 100 kg N ex animal (Table 11).
[g] Land use includes both the area at the farm and the area used to grow the imported feed.

from 55 to 79 g $CO_2/m^2$. According to Cederberg et al. (2013), when ecosystems based on perennial vegetation such as pasture are converted into annual crops, this leads to loss of soil organic carbon and thus $CO_2$ emissions. Both Vleeshouwers and Verhagen (2002) and Vellinga et al. (2004) assumed that growing any type of crop apart from grass would cause a release of C from soil, and they, respectively, assumed this C release would amount to 308 and 11 g $CO_2/m^2$/year. If we assume the use of manure in our calculations, only maize and barley for silage and barley with straw removed would cause a net C release. Differences in time perspective could be part of the explanation for the differences in levels found. In the studies that included soil C sequestration, the time horizon used was often less than the 100 years typically used for other emissions in an LCA.

Another important and uncertain factor for calculating soil C sequestration is data on the input of carbon to soil. Data on below-ground crop residues are very rare and difficult to get hold of as it is very time-consuming and expensive to carry out root wash in studies. In the present paper, it was possible to get data on BG crop residues for Danish conditions, whereas for grazed grass the amount of above-ground crop residues left in the field had to be based on assumption.

A major challenge for a more general use of the suggested method for calculating the contribution from soil carbon changes is whether the method would be valid for showing C input as the difference to that of a reference crop. Here we used 'wheat with no straw removed and no input of manure' as the reference crop and we assumed that it corresponds to no soil carbon changes. That this is an acceptable assumption is illustrated by the good compliance between a calculated overall average release of 28 kg C/ha in Denmark if we upscale our finding for C sequestration/ha for different crops to the average composition of crops grown in Denmark and at the same time assume 80 N input from manure (Statistics Denmark, 2012) and the measured overall Danish average change in soil C of 0 kg C/ha/year (Heidmann et al., 2001).

## 5. Conclusion and perspective for feed supply at farm level

The present study suggests how GHG emissions from a dairy system can be calculated as the sum of GHG contributions from feed production and handling of manure. When combined with methane emissions, the total GHG emissions from the dairy system can be calculated. It is important to consider the contribution from soil carbon storage or loss potential of different land uses and manure systems. The present study illustrated how these elements can be included for different dairy systems.

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