INTERPRETIVE SUMMARY

A case study of the carbon footprint of milk from high performing confinement and grass-based dairy farms. By O'Brien et al., this evaluation of the carbon footprint of high performance dairy systems showed that a grass-based dairy system had a lower carbon footprint per unit of milk compared to confinement dairy systems. However, the ranking of the carbon footprint of high performance grass-based and confinement dairy systems was affected by life cycle assessment (LCA) methodologies, particularly carbon sequestration by grassland. Therefore, a uniform LCA methodology needs to be agreed to assess the carbon footprint per unit of milk from dairy systems.

A case study of the carbon footprint of milk from high performing confinement and grass-based dairy farms

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ABSTRACT

Life cycle assessment (LCA) is the preferred methodology to assess carbon footprint per unit of milk. The objective of this case study was to apply a LCA method to compare carbon footprints of high performance confinement and grass-based dairy farms. Physical performance data from research herds were used to quantify carbon footprints of a high performance Irish grass-based dairy system and a top performing UK confinement dairy system. For the USA confinement dairy system, data from the top 5% of herds of a national database were used. Life cycle assessment was applied using the same dairy farm greenhouse gas (GHG) model for all dairy systems. The model estimated all on and off-farm GHG sources associated with dairy production until milk is sold from the farm in kg of carbon dioxide equivalents (CO_2-eq) and allocated emissions between milk and meat. The carbon footprint of milk was calculated by expressing the GHG emissions attributed to milk per t of energy corrected milk (ECM). The comparison showed when GHG emissions were only attributed to milk, the carbon footprint of milk from the IRE grass-based system (837 kg of CO_2-eq/t of ECM) was 5% lower than the UK confinement system (877 kg of CO_2-eq/t of ECM) and 7% lower than the USA confinement system (898 kg of CO_2-eq/t of ECM). However, without grassland carbon sequestration, the grass-based and confinement dairy systems had similar carbon footprints per t of ECM. Emission algorithms and allocation of GHG emissions between milk and meat also affected the relative difference and order of dairy system carbon footprints. For instance, depending on the method chosen to allocate emissions between milk and meat, the relative difference between the carbon footprints of grass-based and confinement dairy systems varied by 2-22%. This indicates that further harmonization of several aspects of the LCA methodology is required to compare carbon footprints of contrasting dairy systems. In comparison to recent reports that assess the carbon footprint of milk from average Irish, UK and USA dairy systems, this case study indicates that top performing herds of the respective nations have carbon footprints 27-32% lower than average dairy systems. Although, differences between studies are partly explained by methodological inconsistency, the comparison suggests that there is potential to reduce the carbon footprint of milk in each of the nations by implementing practices that improve productivity.

Keywords: carbon footprint, grass, confinement, milk production

INTRODUCTION

A fundamental objective of milk production is to generate sufficient net farm income for dairy farmers (VandeHaar and Pierre, 2006). To achieve this goal in many parts of the developed world, for instance North America, continental Europe and increasingly in the UK, dairy producers aim to increase farm revenue by maximizing milk yield per cow. This is typically accomplished by offering cows nutritionally precise diets in confinement and through improving genetic merit (Arsenault et al., 2009; Capper et al., 2009). Conversely, in some developed countries, notably Ireland and New Zealand, dairy farmers aim to increase profits by minimizing production costs through maximizing the proportion of grazed grass in the diet of lactating cows (Shalloo et al., 2004; Basset-Mens et al., 2009).

Optimizing resource use has the potential to maximize the profitability of grass-based and confinement dairy systems, and improves the environmental sustainability of milk production (Capper et al., 2009). Thus, there is a link between economic performance and environmental sustainability. In recent years, there has been an increasing focus on evaluating the environmental effects of milk production systems, particularly in relation to greenhouse gas (**GHG**) emissions (Thomassen, et al., 2008; Flysjö et al., 2011b). Dairy production is an important source of the dominant GHG emissions, methane (**CH_4**), nitrous oxide (**N_2O**) and carbon dioxide (**CO_2**). Globally, milk production generates 2.7% of GHG emissions with a further 1.3% caused by meat produced from the dairy herd (Gerber et al., 2010). Recent studies suggest that annual global GHG emissions will have to be cut by up to 80% (relative to 1990 levels) before 2050 in order to prevent the worst effects of climate change (Fisher et al., 2007). However, demand for milk products is projected to double between 2000 and 2050 (Gerber et al., 2010). Thus, reducing GHG emissions (carbon footprint) per unit of milk is becoming a necessity for milk producers.

To assess the carbon footprint of milk from contrasting dairy systems, it is necessary to adopt a life cycle approach. This approach, generally referred to as life cycle assessment (**LCA**), entails quantifying GHG emissions generated from all stages associated with a product, from raw-material extraction through production, use, recycling and disposal within the system boundaries (ISO, 2006a,b). Several studies have applied LCA methods to compare carbon footprints of milk from confinement and grass-based dairy farms (Flysjö et al., 2011b; Belflower et al., 2012; O'Brien et al., 2012). However, the results of these studies have been inconsistent.

This inconsistency may be due in part to differences in how GHG emissions are calculated and LCA modeling choices (Flysjö et al., 2011a), but it is also partly due to the farms chosen to represent confinement and grass-based dairy farms. For instance, O'Brien et al. (2012) reported the carbon footprint of milk from a high performing grass-based dairy system was lower than a confinement dairy system exhibiting moderate performance. Conversely, Belflower et al. (2012) showed that the carbon footprint of milk from a commercial confinement dairy system with a noted record of environmental stewardship was lower than a recently established grass-based system. Generally, LCA studies not biased by the farms selected to represent grass-based and confinement dairy systems have reported that grass-based systems produce milk with a lower carbon footprint (Leip et al., 2010; Flysjö et al., 2011b). However, such studies have only considered average performing dairy systems. Thus, there is a need to evaluate the carbon footprint of high performing dairy systems operated at research and commercial farm levels to determine the direction the industry should take to fulfill production and GHG requirements, and to assess their impact on other aspects of the environment such as fossil fuel depletion and land occupation.

In this study, the primary objective was to compare the carbon footprints of milk from high performing confinement and intensive grass-based dairy systems using LCA. To achieve this goal, case study farms located in regions accustomed to grass and confinement based milk production were selected, namely the USA and UK for confinement dairy systems and Ireland for grass-based milk production. A secondary goal of this study was to assess the effect different LCA modeling methodologies have on the carbon footprints of these contrasting milk production systems.

MATERIALS AND METHODS

Description of Dairy Farming Systems

This study used data from existing reports, published studies and databases and required no approval from an animal care and use committee. Physical data (Table 1) for quantifying carbon footprints of milk from the Irish (**IRE**) grass-based dairy system and UK confinement dairy system were obtained from research studies (McCarthy et al., 2007; Garnsworthy et al., 2012). The data used for the IRE dairy system was based on a study carried out to analyze the effect of stocking rate and genetic potential of cows on various biological and economic components of grass-based farms from 2002-2005. The IRE system fed less concentrate than the average or upper quartile of commercial Irish farms in 2011 (590-850 kg DM/cow; Hennessey et al., 2012) and outperformed the top quartile of farms for key technical measures such as milk yield (5,914 kg/cow per year) and milk composition (4.1% fat, 3.5% protein).

The data used for the UK dairy system was based on a study used partly to assess enteric CH_4 emissions from cows in 2010-2011 (Garnsworthy et al., 2012). The technical performance of the UK system was high compared to the upper quartile of commercial herds in the UK in 2011 for milk yield (8,850 kg/cow per year). However, the UK system fed more concentrate than the average or top quartile of farms (2,666-2,684 kg DM/cow; McHoul et al., 2012), but produced more milk per kg of concentrate. Physical data for the USA confinement dairy system was obtained from the DairyMetrics database (DRMS, 2011), and represented the top 5% of herds in 2010-2011 for key technical indicators e.g. milk yield/cow per year.

Irish Grass-Based Dairy System. Milk production in Ireland is based mainly on seasonal-calving grass-based dairy systems. Therefore, the objective of the IRE dairy system was to maximize utilization of grazed grass in the diet of lactating dairy cows. This was accomplished through a combination of extended grazing (early February to late November), tight calving patterns in early spring and rotational grazing of pasture (Dillon et al., 1995). Grass silage was harvested in the IRE dairy system when grass growth exceeded herd feed demand, and fed during the housing period with supplementary minerals and vitamins. Overall, the Irish system was self-sufficient for farm-produced forage. Concentrate feed was purchased onto the farm and offered to cows at the beginning and end of lactation when forage intake was not sufficient to meet nutritional requirements. The total quantity of concentrate offered was 320 kg of DM per cow. Concentrate was given to cows in equal feeds during morning and evening milking. Cows were milked in a 14-unit herringbone milking parlor. The stocking rate of the system was 2.53 livestock units (**LU**; equivalent to 550 kg BW) per ha (McCarthy et al., 2007).

Replacement heifers were raised on-farm in the IRE dairy system and produced their first calf on average at 24 months of age. Heifers primarily grazed pasture, but between November and March, heifers were mainly offered grass silage indoors. Bull calves were sold as early as possible (<3 weeks) in the IRE dairy system. Replacement and cull rates were 18%. The genetics of cows in the IRE dairy system were Holstein-Friesian of New Zealand origin, which were selected over many generations from animals grazing pasture. The genetic potential of the New Zealand Holstein Friesian for each trait of economic importance has been reported (McCarthy et al., 2007). Average calving interval in the IRE dairy system was 368 days and average annual milk yield per cow was 6,262 kg. The on-farm synthetic N fertilizer input in the IRE dairy system was 250 kg N/on-farm ha. Manure produced on-farm was used for on-farm forage production. The majority of manure was deposited by grazing cattle on pasture. Manure was stored as slurry in tanks during the housing period and spread on grassland mainly in spring.

UK and USA Confinement Dairy Systems. Dairy systems increasingly in the UK and USA are based on total mixed ration (TMR) or partial mixed ration (PMR) diets where Holstein-Friesian cows typically produce milk all year round. Thus, in the UK and USA dairy systems cows calved throughout the year, were housed full time and fed TMR or PMR. In the UK dairy system cows were milked individually at automatic (robotic) milking stations. The diet offered was based on data from a UK research herd (Garnsworthy et al., 2012) where cows had ad libitum access to PMR, and concentrates were given to cows during milking. In the USA dairy system it was assumed that cows were milked in an 18-unit herringbone parlor. The composition of the TMR in the USA system was from the survey of Mowrey and Spain (1999), which identified corn silage, alfalfa hay, dry ground corn grain and soybean meal as the typical feedstuffs used in USA dairy production. Diets fed in the UK and USA dairy systems (Table 2) were formulated to fulfill nutrient requirements and maximize production. The chemical composition of the TMR diets offered were similar to previous studies (Kolver and Muller, 1998: Grainger et al., 2009). Maize, grass and whole crop cereal silages were grown on-farm in the UK dairy system. Alfalfa hay and maize silage were assumed to be grown on-farm in the USA dairy system. The remaining feed in both systems was treated as purchased feed. The origin of purchased feed used in the UK, USA and IRE dairy systems was based on trade flow data from the FAO (FAOSTAT, 2012).

Replacement heifers were raised on-farm and produced their first calf on average at 24 months of age (Garnsworthy et al., 2012) in the UK dairy system and 26 months of age in the USA dairy system (DRMS, 2011). Heifers were primarily fed TMR diets in both systems and bull calves were sold within 1 week. The replacement rate in the UK dairy system was 41% and the cull rate 34%. The discrepancy is because the UK herd was expanding. However, to standardize the comparison between dairy systems, the UK herd was assumed to be static (34%). In the US dairy system, the replacement and cull rate was 38%. The genetics of Holstein-Friesian cows in the UK and USA dairy systems were of North American origin (DRMS, 2011; Garnsworthy et al., 2012), which were selected based on generations of animals accustomed to TMR feeding.

Average calving interval in the UK dairy system was 404 d (Garnsworthy et al., 2012) and in the USA dairy system 417 d (DRMS, 2011). Average annual milk yield per cow in the UK dairy system was 10,892 kg (Garnsworthy et al., 2012) and in the USA dairy system 12,506 kg (DRMS, 2011). The on-farm N fertilizer usage in the UK dairy system was 106 kg N/on-farm ha and in the USA dairy system 53 kg N/on-farm ha. Manure produced on-farm was recycled for forage production in the USA dairy system. Approximately, 33% of manure produced on-farm in the UK dairy system was exported and the remainder used for on-farm forage production. Manure from all animals was stored as slurry in the UK dairy system. In the USA dairy system, manure from replacements was stored in a dry lot system and manure from cows was stored in a slurry system.

Greenhouse Gas Modeling

To make the IRE, UK and USA dairy systems as comparable as possible, GHG emissions were calculated using the same dairy farm GHG model (O'Brien et al., 2011, 2012). The GHG model estimates all known GHG emissions from dairy production: CO_2, CH_4, N_2O, and fluorinated gases (**F-gases**). The model uses "cradle to gate" LCA to quantify all on and off-farm GHG sources (e.g. fertilizer, pesticide and fuel manufacture) associated with milk production up to the farm gate. The GHG model operates in combination with Moorepark Dairy System Model (**MDSM**; Shalloo et al., 2004). The MDSM is a whole farm simulation model, which provides input data (animal inventory, feed intakes etc…) for the GHG model. The MDSM uses the net energy (**NE**) and metabolizable energy (**ME**) systems to determine feed requirements (Jarrige, 1989; AFRC, 1993). Calculated feed requirements were validated using actual intake data from the IRE and UK research herds (Horan et al., 2004, 2005; Garnsworthy et al., 2012) and literature reports of typical intakes for high producing USA dairy cows (Wu and Satter, 2000; VandeHaar and Pierre, 2006).

The GHG model calculates on and off-farm GHG emissions by combining farm input data from the MDSM with literature GHG emission algorithms (Tables 3-4). On-farm emission algorithms for CH_4, N_2O and CO_2 emissions from sources such as manure storage and crop residues were obtained from Intergovernmental Panel on Climate Change (**IPCC**) guidelines (IPCC, 2006). However, enteric CH_4 emissions were calculated using country specific approaches (Brown et al., 2012; Duffy et al., 2012; US EPA, 2012). Furthermore, unlike the IPCC (2006) guidelines, gross energy intake (**GEI**) used to calculate enteric CH_4 emissions excluded GEI from rumen protected fat supplements e.g. calcium salts, because, they are not fermentable. On-farm emissions of CO_2 were limited to fossil fuel combustion, urea and lime application. Short-term biogenic sources and sinks of CO_2 such as animals, crops and manure were considered to be neutral with respect to GHG emissions given that the IPCC (2006) and International Dairy Federation (**IDF**, 2010) guidelines assume all carbon absorbed by animals, crops and manure to be quickly released back to the atmosphere through respiration, burning and decomposition

In addition to animals, crops and manure, soils also have the potential to emit or sequester CO_2. Agricultural soils typically lose carbon following the conversion of land to cropland, but gain carbon during the conversion of cropland to grassland. The rate of soil carbon loss or increase declines over time and typically ceases after 20 years once a new soil carbon equilibrium is reached (Rotz et al., 2010). Over the past few decades there has been a decline in the grassland area in the regions analyzed, but this area has not been converted to cropland, which has also declined in area (Brown et al. 2012; Duffy et al. 2012; US EPA, 2012). Thus, the agricultural soils in the USA, UK and Ireland were assumed not to emit CO_2.

Generally, most studies report that soils have a limited capacity to store carbon (Jones and Donnelly, 2004), but recent reports suggest that managed permanent grasslands soils are an important long-term carbon sink (Soussana et al., 2007, 2010). Thus, we also tested the effect of including carbon sequestration. According to the reviews of Conant et al. (2001), Janssens et al. (2005) and Soussana et al. (2010) carbon sequestration rates for permanent Irish, UK and USA grassland soils vary from 0.79-1.74 t/CO_2 per ha per year, partly due to management practices. However, to compare dairy systems, we used the average annual value of these studies (1.19 t/CO_2 per ha) to estimate carbon sequestration by grassland soil.

Off-farm GHG emissions associated with production and supply of non-agricultural products (e.g. pesticide manufacture) were estimated using emission factors from the Ecoinvent database and data from literature sources (Table 4). Emission factors for on-farm sources and purchased non-agricultural products were used in combination with physical data from national statistics (CSO, 2011; Defra, 2011a; USDA, 2011), national reports (Lalor et al., 2010; Defra, 2011b, USDA-NASS, 2011), Ecoinvent and literature reports (Jungbluth et al., 2007; Capper et al., 2009; Vellinga et al., 2012) to quantify emission factors for growing and harvesting purchased feedstuffs. Emissions from processing and transporting feedstuffs were estimated using emission factors from Ecoinvent (2010) and Vellinga et al. (2012). Average sea, rail and road transportation distances and load factors were estimated based on Searates (2012), Jungbluth et al. (2007) and Nemecek and Kägi (2007). Emission factors for importing feedstuffs were estimated by summing emission factors for the farm, processing and transportation stages (Table 4).

Emissions from land use change were estimated for South American soybean and Malaysian palm fruit. The approach used to calculate land use change emissions from these crops was taken from Jungbluth et al. (2007) and involved dividing the total land use change emissions for a crop by the total crop area to estimate the average land use change emissions per crop. This resulted in average land use change emissions per ha from South American soybean of 2.6 t of CO_2 and Malaysian palm fruit 5.5 t of CO_2. For Megalac, which is a calcium salt of palm fatty acid, land use change emissions were not included. This was because the feedstuff is reported to be produced from existing palm forest plantations that do not cause land use change emissions from deforestation (Volac, 2011).

Outputs of the dairy farm GHG model were a static account of annual on-farm and total (on and off-farm) GHG emissions in CO_2 equivalents (**CO_2-eq**). The IPCC (2007) global warming potentials (**GWP**) were used to convert GHG emissions into kg of CO_2-eq using a 100-yr time horizon, where the GWP of CO_2 = 1, CH_4 = 25, and N_2O = 298. The GHG model expresses total GHG emissions as the carbon footprint of milk in kg of CO_2-eq per t of energy corrected milk (**ECM**), which per kg of milk is equivalent to 4% milk fat and 3.3% milk protein (Sjaunja et al., 1990).

Co-product Allocation

Besides producing milk, dairy farms may also export crops, manure and produce meat from culled cows, male calves and surplus female calves. Thus, the carbon footprint of dairy systems should be distributed between these outputs. There is a multitude of methods recommended by various LCA and carbon footprint guidelines to allocate GHG emissions among the co-products of multifunctional systems (ISO, 2006a; IDF, 2010; BSI, 2011). The dairy farm GHG model applies different allocation approaches based on the various guidelines and previous LCA studies of milk.

Allocation of GHG emissions to exported crops was avoided by delimiting the dairy farm GHG model to consider only emissions from crops grown for dairy cattle reared on-farm. The system expansion method recommended by the IDF (2010) LCA guidelines was followed to attribute emissions to exported manure. The method assumes exported manure displaces synthetic fertilizer emissions, but allocates no storage emissions to exported manure. There are several methods to distribute GHG emissions between milk and meat. The following allocation methods were evaluated:

1) Milk – No allocation to meat all GHG emissions attributed to milk.
2) Mass – The GHG emissions of the dairy system was attributed between co-products according to the mass of milk and meat sold.
3) Economic – Allocation of GHG emissions between milk and meat was based on revenue received for milk and meat (sales of culled cows and surplus calves). Prices of milk and animal outputs were estimated using the 2006-2010 market average (CSO, 2011; Defra, 2011a; USDA, 2011).
4) Protein – Edible protein in milk and meat was used to allocate GHG emissions. The protein content of milk was estimated based on Table 1 and the protein content of meat was assumed to be 20% of carcass weight equivalent (**CW**; Flysjö et al., 2011a).
5) Biological – The GHG emissions of the dairy system was allocated based on feed energy required for producing milk and meat. The IDF (2010) guidelines and the MDSM (Shalloo et al., 2004) were used to estimate feed energy required to produce milk and meat.
6) Emission – The GHG emissions associated with producing surplus calves, dairy females <24 months and finishing culled cows were allocated to meat with the remaining emissions assigned to milk (O'Brien et al., 2010; Dollé et al., 2011).
7) System expansion – This approach assumes that meat from culled cows and surplus dairy calves reared for meat replaces meat from alternative meat production systems (Flysjö et al., 2012). In general, meat from traditional cow-calf beef systems is considered as the alternative method of producing meat from a dairy system. The first step of the approach uses LCA to estimate GHG emissions from surplus dairy calves raised for meat and was calculated using the emission

factors of the GHG model where relevant (Tables 3-4) and physical data from Teagasc (2010) for IRE system, Williams et al. (2006) for UK system and Capper et al. (2011) for USA system. The GHG emissions from reared surplus dairy calves were then added to the dairy systems GHG emissions. Subsequently, the meat produced by culled cows and surplus calves raised for meat was summed to estimate the total quantity of meat produced from the dairy system, which was multiplied by the average GHG emissions per kg of meat from cow-calf beef systems. This estimates the displaced GHG emissions from traditional cow-calf meat production and was subtracted from the emissions generated by the dairy systems cows, replacements and surplus dairy calves reared for beef to estimate GHG emissions per unit of milk. The GHG emissions per kg of meat from traditional cow-calf beef systems were calculated according to LCA using the emission factors of the GHG model where applicable and using physical data and emission factors from Foley et al. (2011) for IRE system, Williams et al. (2006) for UK system and Capper et al. (2011) for USA system.

Allocation of GHG emissions was also required for concentrate feeds that are co-products e.g. maize gluten feed. The economic allocation procedure described by IDF (2010) LCA guidelines was used to allocate GHG emissions between concentrate co-products. National reports, Vellinga et al. (2012) and Ecoinvent reports (Jungbluth et al., 2007; Nemecek and Kägi, 2007) were used to estimate concentrate co-product yields and average prices.

Scenario modeling

To assess variability in the emission algorithms of the base dairy farm system described (Tables 3 and 4), the carbon footprint per unit of milk was tested via scenario modeling. The following scenarios were tested relative to the base dairy farm system or baseline scenario:

- Scenario 1 (**S1**): Enteric CH_4 emissions of all dairy systems in S1 were estimated according to the default IPCC (2006) guidelines, which estimates enteric CH_4 emissions as 6.5% of GEI and includes GEI of fat supplements. The remaining emissions sources were estimated using the same algorithms as the baseline scenario.
- Scenario 2 (**S2**): Emission algorithms from the IPCC (1997) guidelines and IPCC (2000) good practice guidelines were applied to estimate emissions from on and off-farm agricultural activities (Supplementary Table 1). Emissions from non-agricultural activities e.g. pesticide manufacture were estimated using the same emissions factors as the baseline scenario (Table 4).
- Scenario 3 (**S3**): Country specific emission algorithms from national GHG inventories (Brown et al. 2012; Duffy et al. 2012; US EPA, 2012) and literature sources were used to estimate emissions from on and off-farm agricultural activities (Supplementary Tables 2 and 3). Emissions from non-agricultural activities were estimated using national literature sources (Table 4).

RESULTS

On-farm GHG emissions and carbon footprint of milk from dairy systems

Table 5 shows GHG profiles, on-farm GHG emissions and carbon footprints (on and off-farm GHG emissions) per t of ECM, with no allocation of GHG emissions to meat, for the IRE, UK and USA dairy systems. On-farm GHG emissions per t of ECM was lowest for the UK confinement dairy system, was 13% greater for the IRE grass-based dairy system, and was 15% greater for the USA confinement dairy system. Carbon footprint per t of ECM was lowest for the IRE grass-based dairy system, was 5% greater for the UK confinement dairy system, and was 7% greater for the USA confinement dairy system.

The GHG profiles of Table 5 show that the main sources of GHG emissions from the IRE dairy system were enteric CH_4 (47%), N_2O emissions from manure deposited on pasture by grazing cattle (15%), CO_2 and N_2O emissions from fertilizer application (12%), GHG emissions from fertilizer production (8%), and CH_4 and N_2O emissions from manure storage and spreading (8%). The key sources of GHG emissions from the UK dairy system were enteric CH_4 (42%), CH_4 emissions from manure storage (13%), GHG emissions from imported concentrate feed (12%), N_2O emissions from manure storage and spreading (9%), CO_2 emissions from electricity generation and fuel combustion (7%) and CO_2 emissions from land use change (6%). The main sources of GHG emissions from the USA dairy system were enteric CH_4 (42%), N_2O emissions from manure storage and spreading (17%), CH_4 emissions from manure storage (14%), GHG emissions from imported concentrate feed (12%), and CO_2 emissions from electricity generation and fuel combustion (8%).

The GHG profiles also show that sequestration by grassland soil had no effect or a minor mitigating effect on GHG emissions of the UK and USA dairy systems (0-2%), but had a large effect on the IRE dairy system (9%). Thus, excluding carbon sequestration affected the ranking and relative difference between dairy systems in on-farm GHG emissions and carbon footprint per t of ECM. The analysis showed that when carbon sequestration was excluded on-farm GHG emissions per t of ECM was lowest for the UK confinement dairy system, was 12% greater for the USA confinement dairy system, and was 22% greater for the IRE grass-based dairy system. Excluding carbon sequestration, resulted in the confinement and grass-based dairy systems emitting a similar carbon footprint per t of ECM.

Allocation of GHG emissions between milk and meat

The effects of using different methods to allocate GHG emissions between milk and meat on the carbon footprint per t of ECM for the IRE, UK and USA dairy systems are shown in Figure 1. Within the dairy systems there was a difference of up to 41% in the proportion of dairy system GHG emissions that were allocated to milk depending on the methodology used. Excluding attributing all GHG emissions to milk, mass allocation attributed the most GHG emissions to milk followed by protein, economic, biological, emissions allocation and system expansion.

The comparison of allocation methods shows that mass and protein allocation attributed a fixed proportion of GHG emissions to milk for each dairy system, 98% and 94%, respectively. Thus, the ranking and relative difference between dairy systems carbon footprint per t of ECM was unchanged compared to attributing no GHG emissions to meat. The proportion of GHG emissions allocated to the carbon footprint of ECM varied between dairy systems for economic, biological and emission allocation methods. For instance, allocation on an emission basis attributed 85% of GHG emissions to milk for IRE dairy system, 84% for UK dairy system and 81% for USA dairy system. This resulted in the UK dairy system, instead of the USA dairy system, having the highest carbon footprint per t of ECM. Thus, the ranking of dairy systems' carbon footprint per t of ECM was inconsistent between allocation methods.

System expansion did not affect the ranking of dairy systems carbon footprint per t of ECM, but the approach led to a significantly greater relative difference between the carbon footprints of grass-based and confinement dairy systems compared to the other allocation methods analyzed. The approach showed that the IRE grass-based system had a carbon footprint per t of ECM 18% lower than the UK confinement system and 22% lower than the USA confinement dairy system.

Scenario analysis

The results of S1 (Table 5) showed that applying the general IPCC (2006) guidelines to estimate enteric CH_4 emissions as 6.5% of GEI (with GEI from fat supplements included) increased carbon footprints per t of ECM of the confinement dairy systems by 4-5% compared to the baseline scenario. However, using this approach to estimate enteric CH_4 emissions had little effect on carbon footprint per t of ECM (<1%) of the grass-based dairy system, because enteric CH_4 emissions was estimated as 6.45% of GEI in the baseline scenario. Thus, the relative difference between grass-based and confinement dairy systems carbon footprint per t of ECM was greater in S1 than the baseline scenario.

Under S2, the original IPCC (1997, 2000) emission algorithms for agricultural sources increased estimates of CH_4 emissions from manure storage, GHG emissions from concentrate production, and N_2O emissions from manure and fertilizer compared to the baseline scenario. The increase in N_2O emissions from on-farm fertilizer use was greater for the grass-based dairy system than for the confinement dairy systems in S2. However, the increase in CH_4 emissions from manure storage and GHG emissions from concentrate production was greater for the confinement dairy systems, than for the grass-based dairy system. In addition, S2 increased enteric CH_4 emissions from the confinement dairy systems, but had the opposite effect on the grass-based dairy system. As a result, S2 caused a greater increase in the carbon footprints per t of ECM of the confinement dairy systems (24-28%) than the grass-based dairy system (13%) relative to the baseline scenario.

The country specific emission algorithms of S3 reduced N_2O emissions from manure excreted by grazing cattle, and CH_4 and N_2O emissions from manure storage and spreading relative to the baseline. In addition, S3 estimated lower GHG emissions from concentrate and fertilizer production for the USA dairy system. However, the scenario had no effect or increased emissions from concentrate and on-farm fertilizer use for the IRE and UK dairy systems. This resulted in the country specific emission algorithms of S3 reducing the carbon footprint of the UK dairy system by 4% relative to the baseline, but by 9-10% for the IRE and USA dairy systems. Consequently, the order of carbon footprints per t of ECM of dairy systems in S3 was not consistent with the baseline scenario.

DISCUSSION

Life cycle assessment studies that directly compare carbon footprints of milk from high performance grass-based and confinement dairy systems within or across countries are rare. The direct comparison in this study therefore provided a unique opportunity to evaluate the effect that contrasting high performance dairy systems have on the carbon footprint of milk and individual GHG sources. The results implied that high performance grass-based systems are capable of having a lower carbon footprint per unit of milk compared to top performing confinement dairy systems. However, this difference was principally due to the inclusion of carbon sequestration, which confers a degree of uncertainty upon the conclusions due to the lack of solid sequestration data available. The ranking of the carbon footprint of milk from high performance grass-based and confinement dairy systems was also influenced by LCA modeling choices e.g. allocation methods and emissions algorithms. This agrees with the outcomes of previous research (Flysjö et al., 2011a; O'Brien et al., 2011, 2012; Zehetmeier et al., 2012) and implies there is a need to agree a uniform LCA methodology for milk production. It is also important to emphasize that all physical data used in this study were a snapshot in time and changes in feeding systems and performance could alter the conclusions.

Comparison of GHG emissions and carbon footprint of milk from high performance grass-based and confinement dairy systems

In agreement with previous studies (Leip et al., 2010; Flysjö et al., 2011b; Belflower et al., 2012), the main source of GHG emissions, enteric CH_4, was greater per LU from the confinement dairy systems than the grass-based dairy system, but lower per unit of milk. The greater milk yield per cow and higher replacement rate within the confinement systems explained the greater enteric CH_4 emissions per LU, because these factors increase DMI per LU, which is a key determinant of enteric CH_4 emissions (O'Neill et al., 2011). Milk yield per cow was greater in the confinement systems than the grass-based system, given the greater genetic selection for milk yield and increased levels of concentrate feeding. These factors also explained the lower enteric CH_4 emissions per unit of milk of the confinement dairy systems, because concentrate rich diets generally contain less fiber than forage diets and improving genetic merit increases productivity, which facilitates the dilution of maintenance effect whereby the resource cost per unit of milk is reduced (Capper et al., 2009).

Previous modeling research by Garnsworthy (2004) agrees with this analysis that increasing milk yield reduces enteric CH_4 emissions per unit of milk and showed that at similar annual milk yields, improving the fertility of dairy cows decreases enteric CH_4 emissions per unit of milk. This was because improving cow fertility reduces the number of replacement heifers required to maintain the herd size for a given milk quota or number of cows, which reduces enteric CH_4 emissions. The results of Garnsworthy (2004) also partially explain why the lower replacement rate of the UK confinement dairy system resulted in similar enteric CH_4 emissions per unit of milk as the USA confinement dairy system, even though annual ECM yield per cow was 10% greater in the USA dairy system.

Another key reason that explained the similar enteric CH_4 emissions per unit of milk of the confinement systems was the different diets fed. Unlike the diet fed in the USA system, the formulation of the diet of cows in the UK system included protected lipids, which compared to forage and most concentrate feeds reduce enteric CH_4 emissions, because protected lipids are not fermentable in the rumen (Martin et al., 2010). In addition, they slightly increased the feed efficiency (kg DM/unit of milk) of the UK dairy system relative to the USA dairy system, which partly led to the UK and USA systems emitting similar enteric CH_4 emissions per unit of milk. However, in contrast to the UK system, the diet of cows in the USA system was formulated based on a national survey of common feedstuffs (Mowrey and Spain, 1999). Thus, the USA diet may not truly reflect high performance systems, which would also explain in part the difference in feed efficiency between confinement dairy systems.

The greater feed efficiency of the UK confinement system also in part reduced GHG emissions from manure storage and on-farm feed production, which resulted in lower on-farm GHG emissions per unit of milk relative to the USA confinement system. This was because feed intake is a key determinant of GHG emissions from these sources (Basset-Mens et al., 2009; Flysjö et al., 2011b). As well as feed intake, the method of storage affects GHG emissions from manure storage (IPCC, 2006). Manure from all animals was managed in a liquid system for the UK confinement system, but for the USA confinement system, manure from replacements was managed in a dry lot. This caused the USA system to emit greater N_2O emissions and therefore greater GHG emissions per unit of milk from manure storage. On-farm GHG emissions per unit of milk were also greater from the USA system relative to the UK system, because the USA system recycled all manure on-farm to produce forage for ruminants, but the UK system exported a third of manure produced in order to stay within European regulations for slurry application in a nitrate vulnerable zone. Furthermore, the manure exported from the UK system was assumed to displace synthetic fertilizer (IDF, 2010), which further reduced on-farm GHG emissions.

Compared to the IRE grass-based dairy system the UK and USA confinement dairy systems were more feed and N efficient, but also fed more conserved forages. Thus, the confinement dairy systems harvested more feed mechanically and, albeit based on inconsistent research (Jones and Donnelly, 2004), sequestered less carbon compared to the IRE grass-based dairy system, because the majority of forage was grown on arable land. As a result, on-farm GHG emissions per unit of milk of the IRE grass-based dairy system were lower than the USA confinement dairy system. However, the feed efficiency and carbon sequestration of the UK confinement system was greater than the USA confinement system. This led to the UK confinement dairy system emitting the lowest on-farm GHG emissions per unit of milk.

Consistent with previous reports (Belflower et al., 2012; O'Brien et al., 2012), GHG emissions from production and transport of purchased concentrate feed, manufacture of fertilizer for on-farm feed production and from electricity generation were the main contributors to dairy systems off-farm GHG emissions. The IRE grass-based system emitted the lowest off-farm GHG emissions per unit of milk, which can be explained by the low reliance of the grass-based system on purchased concentrate (O'Brien et al., 2012). Off-farm GHG emissions per unit of milk were greater from the UK confinement system than the USA confinement system, given the greater feeding of concentrate feeds associated with a high GHG emission (e.g. South American soybeans) in the UK system. This is similar to the finding of Gerber et al. (2010), who reported that production of South American soybeans used in European dairy systems emits significant CO_2 emissions from deforestation.

The greater off-farm GHG emissions per unit of milk of the UK confinement dairy system led to the UK system emitting a greater carbon footprint than the IRE grass-based dairy system. However, the carbon footprint of the UK confinement dairy system was lower than the USA confinement dairy system, because as discussed, on-farm GHG emissions per unit of milk were greater from the USA system. The lower carbon footprint of milk from the grass-based dairy system compared to the confinement dairy systems agrees with some reports (Leip et al., 2010; Flysjö et al., 2011b; O' Brien et al., 2012) but disagrees with others (Capper et al., 2009; Belflower et al. 2012). This can be explained by the performance of dairy systems compared, but also by the variation in the application of the LCA methodology.

Influence of LCA methodology on the carbon footprint of milk from dairy systems

Major methodological decisions of LCA include the selection of GHG emission algorithms and how to allocate environmental impacts such as GHG emissions between co-products e.g. milk and meat of multifunctional systems. Although international standards (ISO, 2006a; IDF, 2010; BSI, 2011) have been developed for LCA methodology, the standards are not consistent particularly regarding allocation methodologies. Several criteria can be used to allocate GHG emissions between milk and meat e.g. economic value or mass basis. Choosing different methodologies to allocate GHG emissions between milk and meat affects the carbon footprint of milk and can change the ranking of the carbon footprints of milk from dairy systems (Flysjö et al., 2012). For instance, choosing to allocate dairy system GHG emissions between milk and meat on a mass basis for the UK confinement dairy system, but on an economic basis for the USA confinement dairy system, resulted in the UK system having a greater carbon footprint per t of ECM than the USA system. However, when mass or economic allocation was used for both dairy systems, the UK system had a slightly lower carbon footprint per t of ECM. Thus, to facilitate a valid comparison of the carbon footprints of milk from different dairy systems the same method must be used to allocate GHG emissions between milk and meat.

Similar to previous studies (Cederberg and Stadig, 2003; O'Brien et al., 2012), allocation according to physical relationships such as mass, protein content or economic value resulted in a greater carbon footprint per unit of milk relative to allocation based on physical causal relationships (e.g. biological energy required to produce milk and meat from dairy cows and surplus calves). The differences between these allocation methods was explained by the relatively high energy requirements of producing meat from dairy systems compared to the mass or economic value of meat produced. The assessment of allocation methods showed, similar to Flysjö et al. (2011a), that even when the same allocation method was applied the percentage of GHG emissions allocated between milk and meat varied depending on dairy system. As a result, the ranking of carbon footprints of milk from dairy systems was not consistent between allocation methods. Thus, for a given dairy system there are advantages and disadvantages to choosing a particular allocation procedure.

Another method evaluated to handle allocation of GHG emissions between co-products was system expansion. Similar to previous studies, the methodology was applied to assume meat from dairy production (including meat from surplus dairy calves raised for finishing) substitute's meat from traditional cow-calf beef systems (Flysjö et al., 2012). This assumption resulted in a large deduction in GHG emissions of dairy systems, because meat production from cow-calf beef systems generates a substantially larger GHG emissions per unit of meat (30-40%) than an equal quantity of meat produced from dairy systems (Williams et al., 2006). Thus, applying this approach resulted in a significantly lower carbon footprint per unit of milk, compared to the other allocation methods. Furthermore, system expansion caused the greatest relative difference between the grass-based and confinement systems carbon footprints per t of ECM. This was because for a fixed farm milk output increasing milk yield per cow generally reduces meat production from dairy system (Cederberg and Stadig, 2003; Flysjö et al. 2012). Thus, the confinement systems displaced less meat per unit of milk from traditional cow-calf beef systems, compared to the grass-based system. Consequently, the deduction in confinement systems GHG emissions per unit of milk was lower than the grass-based system.

In addition to the quantity of meat a dairy system produces, the demand for meat and the type of meat a dairy system substitutes can significantly affect the carbon footprint of milk using system expansion. For instance, Flysjö et al. (2012) reported that conventional dairy systems had a greater carbon footprint per unit of milk than organic dairy systems when meat from dairy systems was assumed to replace meat from cow-calf beef systems, but conventional systems had the opposite effect when meat from dairy systems was assumed to substitute pork. Thus, this demonstrates that system expansion increases the uncertainty of the carbon footprint of milk from dairy systems compared to allocation based on causal or non-causal relationships. Furthermore, the methodology can create an unfair bias against meat by attributing the production of dairy animals entirely to meat (Rotz et al., 2010). Conversely, some non-causal allocation methods were biased against milk because they attributed little (2%) or no GHG emissions to meat. Thus, this suggests more moderate options e.g. economic or biological allocation are the most suitable methods to distribute GHG emissions between milk and meat.

Aside from allocation methods, LCA choices regarding emission algorithms affect the carbon footprint of milk. For instance, scenario modeling showed that computing GHG emissions with country specific emission algorithms for each nation ranked carbon footprints of milk from dairy systems differently to calculating emissions with the same emission algorithms for all country. Thus, this suggests that where nations differ in their efforts to measure emissions, it is more appropriate, albeit less precise, to use the same computation approach for each region (Flysjö et al., 2011b). However, consistent with previous reports (Basset-Mens et al., 2009; Rotz et al., 2010) relatively few emission algorithms influence the carbon footprints of milk from dairy systems. The algorithms that affected both the grass and confinement systems were enteric CH_4 emission algorithms, N_2O emission factors for manure spreading and emission factors related to fertilizer. Similar to O'Brien et al. (2012), the carbon footprint of milk from the grass-based system was also affected by the N_2O emission factor for manure deposited during grazing given the short housing period (80 d). The N_2O emission factor for manure excreted by grazing cattle, however, had no affect on the carbon footprint of milk from the confinement systems, which were instead influenced by the CH_4 and N_2O emission algorithms for manure storage.

Carbon sequestration and land use change emissions

Evaluations of the carbon footprint of milk from dairy systems are affected by LCA methodological decisions regarding carbon sequestration and land use change emissions from tropical deforestation and increased cropping. For instance, when carbon sequestration was included the grass-based dairy system had the lowest carbon footprint per t of ECM, but omitting sequestration resulted in the grass-based and confinement dairy systems having similar carbon footprints per t of ECM. On the one hand, LCA standards recommend excluding carbon sequestration, because the IPCC (2006) guidelines assume that soil's ability to store carbon reaches equilibrium after a fixed period (20 years). On the other hand, some (e.g. Leip et al., 2010) argue that carbon sequestration should be included given the recent findings of Soussana et al. (2007, 2010) that managed grassland soils can permanently sequester carbon. However, given the uncertainty associated with carbon sequestration by managed permanent grassland, more research and data are required to accurately include sequestration and determine if it causes differences between the carbon footprints of milk from grass-based and confinement dairy systems.

There is also lack of consensus on how to assess land use change emissions. For instance, Gerber et al. (2010) and Leip et al. (2010) assume that the expansion of certain crops in particular regions (e.g. soybean in South America) causes land use change emissions from deforestation. However, others (e.g. Audsley et al., 2009) assume that all land occupation either directly or indirectly causes emissions from land use change. Thus, instead of applying an emission factor for land use change to a particular crop e.g. Brazilian soybean, the approach applies a general emission factor for land use change to all occupation of land. The method suggested by Gerber et al. (2010) and Leip et al. (2010) was followed in this study, but using a different approach, such as a general emission factor for land use change, can alter the order of dairy systems carbon footprints per unit of milk (Flysjö et al., 2012). Thus, there is need to develop a harmonized approach to assess land use change emissions.

Comparison with carbon footprint studies of milk

Results of LCA and carbon footprint studies of milk are difficult and rarely completely valid to compare, because of potentially large differences in the application of the LCA methodology as outlined previously. Nevertheless, differences can be partly explained by inherent differences between dairy systems. In general, the carbon footprint estimates of the high performance IRE grass-based dairy system and top performing UK and USA confinement dairy systems were at the lower end of the range of recent carbon footprint reviews and studies of milk (Crosson et al., 2011; Flysjö et al., 2011a,b; Gerber et al., 2011). Relative to recent national average estimates of carbon footprints of IRE, UK and USA dairy production (Capper et al., 2009; Leip et al., 2010; Thoma et al., 2013), our findings suggest that high performance dairy systems of these countries reduce carbon footprint of milk by 27-32%, however, this comparison is partially affected by methodological differences.

Excluding methodology differences, the lower carbon footprint of milk from high performance dairy systems can be explained by their greater productive efficiency, which potentially reduces resource use per unit of milk, thereby reducing carbon footprint (Capper et al., 2009). Furthermore, comparison of carbon footprints of milk from high performance dairy systems in this study relative to recent reports of carbon footprints of average IRE, UK and USA dairy systems indicates that the relative difference between average and high performance dairy systems was likely to be greater than the relative difference between top performing grass and confinement dairy systems. This is similar to the results of Van der Werf et al. (2009) and suggests that improving productivity of dairy systems has a greater affect on the carbon footprint of milk than converting from a confinement dairy system to an intensive grass-based system or vice versa.

CONCLUSIONS

Comparisons of the carbon footprints per unit of milk from high performing dairy systems showed that the UK and USA confinement dairy systems had a similar carbon footprint, but the Irish grass-based dairy system had a lower carbon footprint per unit of milk when carbon sequestration and direct allocation of land use change emissions were included in the calculations. However, the relative differences and ranking of dairy systems carbon footprints per unit of milk were not consistent in this study when different LCA methodologies regarding, GHG emission algorithms, carbon sequestration and allocation decisions between milk and meat were used. In particular, choosing to exclude carbon sequestration resulted in the grass-based and confinement dairy systems having similar carbon footprints per unit of milk. Therefore, this implies that further harmonization of several aspects of the LCA methodology is required to compare carbon footprints of milk from contrasting dairy systems. This study also indicates that there is significant potential to reduce the carbon footprint of milk in each of the countries by adopting farm practices currently implemented at research level and by top performing commercial milk producers.

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Table 1 Technical description of a high performance Irish grass-based dairy system, a high performance UK confinement system and a top performing USA confinement dairy system

Item	Unit	Irish	UK	USA
On-farm size	ha	40	85	93
Off-farm size[1]	ha	3	97	82
Milking herd	# milking cows	92	220	153
Milk production	kg milk/cow per yr	6,262	10,892	12,506
ECM[2] production	kg ECM/cow per yr	6,695	10,602	11,650
Milk fat	%	4.47	3.95	3.58
Milk protein	%	3.55	3.14	3.17
Calving interval	days	368	404	417
Replacement rate	%	18	34	38
Cull rate	%	18	34	38
Average BW	kg	543	613	680
Stocking rate	LU[3]/ha	2.53	3.74	2.79
Concentrate	kg DM/cow per yr	320	2,905	3,355
Grass[4]	kg DM/cow per yr	4,099	-	-
Alfalfa hay	kg DM/cow per yr	-	-	2,570
Grass silage	kg DM/cow per yr	849	1,142	-
Maize silage	kg DM/cow per yr	-	1,862	2,155
Whole crop wheat silage	kg DM/cow per yr	-	825	-
Rape straw	kg DM/cow per yr	-	219	-
Total intake	kg DM/cow per yr	5,270	6,953	8,079
On-farm N fertilizer	kg N/on-farm ha per yr	250	106	53
Manure exported	%	0	33	0

[1] Off-farm land area required to produce purchased forage and concentrate feedstuffs.

[2] ECM = Energy corrected milk = $(0.25 + 0.122 \times \%\text{fat} + 0.077 \times \%\text{protein}) \times$ kg milk (Sjaunja et al., 1990).

[3] LU = Livestock unit equivalent to 550 kg BW.

[4] Forage intakes were estimated with the Moorepark Dairy System Model (Shalloo et al., 2004) using milk production, animal BW, concentrate supplementation and feed ration composition data.

Table 2 Formulation and composition of diets fed to lactating Holstein-Friesian dairy cows in the UK and USA confinement dairy systems and concentrate offered to lactating Holstein-Friesian cows at pasture for the Irish dairy system.

	UK		USA	Ireland
Item	Jan-May	June-December	Full Year	Jan-March and October-November
Ingredient (g/kg DM)				
Grass silage	132	118	-	-
Maize silage	320	362	250	-
Whole crop wheat silage	126	180	-	-
Alfalfa hay	-	-	305	-
Rape straw	50	27	-	-
Rolled barley	-	-	-	250
Corn grain dry ground	-	-	265	-
Sugar beet pulp	96	-	-	350
Corn gluten	-	-	-	260
Rapeseed meal	132	139	-	-
Soybean meal[1]	84	89	150	110
Molasses		36	-	-
Megalac[2]	23	30	-	-
Minerals and vitamins	37[3]	19[3]	30[4]	30[5]
Composition				
ME (MJ/kg DM)	11.2	11.4	11.4	11.9
CP (g/kg DM)	168	170	182	180
NDF (g/kg DM)	359	278	340	315
Concentrate[6] feeding during robotic milking				
Concentrate per cow (kg/d)	1.6	3.0	-	-
Milk yield threshold for extra concentrate feed (L/d)	31	35	-	-
Kilogram of concentrate per L milk yield above threshold	0.33	0.45	-	-

[1] Based on FAOSTAT (2012), 95% of soybean meal in the UK dairy system was from South America and 5% from USA, for the IRE system 92% of soybean meal was from South America and 8% from the USA, for the USA system all soybean meal was from the domestic market.

[2] Megalac = Calcium salts of palm oil fatty acid distillate. Volac International Ltd., Royston, UK. Palm oil was sourced from sustainable forest plantations in Malaysia.
[3] Calcium 18%, phosphorus 10%, magnesium 5%, salt 17%, copper 5000 mg/kg, manganese 5,000 mg/kg, cobalt 100 mg/kg, zinc 6,000 mg/kg, iodine 500 mg/kg, selenium 25 mg/kg, vitamin A 400,000 IU/kg, vitamin D_3 80,000 IU/kg, and vitamin E 1,000, mg/kg.
[4] Calcium carbonate 33%, dicalcium phosphate 23%, sodium bicarbonate 20%, salt 13%, magnesium oxide 7%, copper 13,350 mg/kg, iron 23,990 mg/kg, manganese 51,000 mg/kg, cobalt 430 mg/kg, zinc 62,010 mg/kg, iodine 1,030 mg/kg, selenium 320 mg/kg, vitamin A 700,000 IU/kg, vitamin D 222,000 IU/kg, and vitamin E 17,600, mg/kg.
[5] Selenium 60 mg/kg, iodine 700 mg/kg, copper 4000 mg/kg, zinc 5000 mg/kg, vitamin A 250,000 IU, vitamin D 50,000 IU, vitamin E 2,000 IU.
[6] Concentrate formulation on a DM basis, citrus pulp 18%, dried distillers grains 17%, soy hulls 16%, rapeseed meal 15%, corn gluten feed 10%, barley 6%, corn grain 5%, molasses 4%, palm kernel meal 4%, vegetable oil 3%, minerals and vitamins 2%.

Table 3 Emission factors used in the baseline scenario of the dairy farm greenhouse gas (GHG) model (O'Brien et al., 2011) for quantification of on-farm GHG emissions

Emission and source	Emission factor	Unit
Methane (CH_4)		
Enteric fermentation[1]		
Dairy cow IRE (housing)	DEI[2] × (0.096 + 0.035 × S_{DMI}[3]/T_{DMI}[4]) - (2.298 × FL[5] – 1)	MJ/d
Dairy cow IRE (grazing)	0.06 × GEI[6]	MJ/d
Heifer IRE	0.065 × GEI	MJ/d
Dairy cow UK	0.06 × GEI	MJ/d
Dairy cow USA	0.055 × GEI	MJ/d
Heifer UK and USA	0.06 × GEI	MJ/d
Manure storage	Manure VS[7] stored × 0.24 × 0.67 × (MS_a[8] × 0.17 + MS_b[9] × 0.02 + MS_c[10] × 0.001 + MS_d[11] × 0.01)	kg/year
Grazing returns[12]	Manure VS excreted on pasture × 0.24 × 0.67 × 0.01	kg/year
Ammonia (NH_3-N)		
Synthetic N fertilizer	0.1 × N fertilizer	kg/kg N
Slurry storage	0.4 × slurry N stored	kg/kg N
Solid manure storage	0.3 × solid manure N stored	kg/kg N
Manure application	0.2 × (N stored – NH_3 storage loss)	kg/kg N
Grazing returns	0.2 × N excreted on pasture	kg/kg N
Nitrate (NO_3^--N)		
N leaching	0.3 × N applied	kg/kg N
Nitrous oxide (N_2O-N)		
Grazing returns	0.02 × N excreted on pasture	kg/kg N
Synthetic N fertilizer	0.01 × N fertilizer	kg/kg N
Manure application	0.01 × (N stored – N storage loss)	kg/kg N
Crop residues	0.01 × N Crop Residues	kg/kg N
Slurry storage	0.005 × slurry N stored	kg/kg N
Solid manure storage	0.005 × solid manure N stored	kg/kg N
Dry lot	0.02 × dry lot manure N stored	kg/kg N
Nitrate leaching	0.0075 × N leached	kg/kg NO_3^--N
Ammonia re-deposition	0.01 × sum of NH_3 emissions	kg/kg NH_3-N
Carbon dioxide (CO_2)		
Diesel	2.63 × diesel use	kg/l
Gasoline	2.30 × gasoline use	kg/l
Kerosene	2.52 × kerosene use	kg/l
Lime	0.44 × lime application	kg/kg lime
Urea	0.73 × urea application	kg/kg urea

[1] Country specific emission factors were used to estimate enteric fermentation methane emissions for the Irish seasonal grass-based dairy system (IRE), UK confinement dairy system (UK) and USA Confinement dairy system (USA). The remaining emission sources were estimated according to the IPCC (2006) guidelines

[2] DEI = Digestible energy intake.

[3] S_{DMI} = Silage dry matter intake.

[4] T_{DMI} = Total dry matter intake.

[5] FL = Feeding levels above maintenance energy.

[6] GEI = Gross energy intake.
[7] VS = Volatile solids.
[8] MS_a = Proportion of manure volatile solids stored in slurry system.
[9] MS_b = Proportion of manure volatile solids stored in solid storage system. Solid manure dry matter content >20%.
[10] MS_c = Proportion of manure volatile solids spread daily.
[11] MS_d = Proportion of manure volatile solids stored in dry lot.
[12] Manure excreted by grazing cattle on pasture.

Table 4 Emissions factors used in the dairy farm greenhouse gas (GHG) model (O'Brien et al., 2011) for quantification of off-farm GHG emissions from manufacture and transport of key purchased inputs in g of CO_2 equivalents[1].

Item	Baseline and Scenario 1[2]	Scenario 2[3]	Scenario 3[4]	References
Electricity Ireland, kWh	612	612	612	Ecoinvent (2010), Howley et al. (2011)
Electricity UK, kWh	612	612	597	Ecoinvent (2010), Defra (2011c)
Electricity USA, kWh	612	612	658	Ecoinvent (2010), Defra (2011c)
Diesel, kg	359	359	359	Ecoinvent (2010)
Gasoline, kg	455	455	455	Ecoinvent (2010)
Kerosene, kg	341	341	341	Ecoinvent (2010)
Ammonium-based fertilizer EU, kg N	5,164	5,164	5,164	Ecoinvent (2010), Leip et al. (2010)
Ammonium-based fertilizer US, kg N	5,164	5,164	3,616	Snyder et al. (2009), Ecoinvent (2010)
Urea EU, kg N	2,627	2,627	2,627	Ecoinvent (2010), Leip et al. (2010)
Urea USA, kg N	2,627	2,627	1,616	Snyder et al. (2009), Ecoinvent (2010)
Lime, kg	43	43	43	Ecoinvent (2010)
P fertilizer, kg P_2O_5	1,926	1,926	1,926	Ecoinvent (2010)
K fertilizer, kg K_2O	363	363	363	Ecoinvent (2010)
Pesticide, kg active ingredient	7,421	7,421	7,421	Ecoinvent (2010)
Milk replacer, kg	1.38	1.42	1.34	Ramírez et al. (2006), Ecoinvent (2010)
Barley, kg DM	373	434	365	Ecoinvent (2010), Vellinga et al. (2012)
Corn grain USA, kg DM	380	455	323	Ecoinvent (2010), Vellinga et al. (2012)
Corn grain Europe, kg DM	412	474	417	Ecoinvent (2010), Vellinga et al. (2012)
Sugar beet pulp[5], kg DM	61	70	57	Ecoinvent (2010), Vellinga et al. (2012)
Corn gluten, kg DM	1,078	1,120	1,061	Ecoinvent (2010), Vellinga et al. (2012)
DDGS[6], kg DM	929	931	927	Ecoinvent (2010), Vellinga et al. (2012)
Rapeseed meal, kg DM	482	591	468	Ecoinvent (2010), Vellinga et al. (2012)
Soyabean meal South America[7], kg DM	1,472	1,664	1,477	Ecoinvent (2010), Vellinga et al. (2012)
Soyabean meal USA, kg DM	299	495	336	Ecoinvent (2010), Vellinga et al. (2012)
Straw, kg DM	41	50	38	Ecoinvent (2010), Vellinga et al. (2012)
Molasses, kg DM	149	169	141	Ecoinvent (2010), Vellinga et al. (2012)

Item	Baseline and Scenario 1[2]	Scenario 2[3]	Scenario 3[4]	References
Megalac[8], kg DM	1,032	1,120	1,020	Ecoinvent (2010), Vellinga et al. (2012)

[1] Carbon dioxide = 1; methane = 25; nitrous oxide = 298 (IPCC, 2007).

[2] The baseline scenario and scenario 1 used emission algorithms from the current IPCC (2006) guidelines to estimate emissions from agricultural GHG sources related to the production of feedstuffs.

[3] Scenario 2 applied the same emission factors as the baseline scenario to estimate emission from non-agricultural products e.g. electricity and used, but applied emission algorithms from the original IPCC (1997) guidelines and IPCC (2000) good practice guidelines to estimate emissions from agricultural GHG sources related to the production of feedstuffs.

[4] Scenario 3 used country specific emission factors to estimate emissions from the manufacture of non-agricultural products and used country specific emission algorithms to estimate emissions from agricultural GHG sources related to the production of feedstuffs.

[5] Emissions were allocated between co-products based on their economic value using national data, Ecoinvent (2010) and Vellinga et al. (2012).

[6] DDGS = Dried distillers grains with solubles.

[7] Based on Ecoinvent (2010), 62% of South American soybean was from Argentina and 38% was from Brazil.

[8] Megalac = Calcium salts of palm oil fatty acid distillate. Volac International Ltd., Royston, UK.

Table 5 Carbon footprints[1] with all greenhouse gas (GHG) emissions attributed to milk of a high performance Irish grass-based dairy system, a high performance confinement UK dairy system and a top performing confinement USA dairy system calculated using a life cycle assessment dairy farm GHG model (O'Brien et al., 2011)

		Baseline[2]			S1[3] % baseline change			S2[4] % baseline change			S3[5] % baseline change		
Emission and source	Location	Irish	UK	USA	Irish	UK	USA	Irish	UK	USA	Irish	UK	USA
Methane (kg CO_2-eq[6]/t ECM[7])													
Enteric fermentation	On-farm	430.69	376.39	373.60	0.8	10.4	11.6	-5.0	2.8	5.5	-	-	-
Manure storage and spreading	On-farm	42.09	118.60	121.91	-	-	-	111.4	129.1	127.1	-16.0	-31.3	-32.7
Fertilizer production	Off-farm	1.61	0.34	0.39	-	-	-	-	-	-	-	-	-12.8
Concentrate production	Off-farm	0.82	2.38	1.55	-	-	-	-	-	-	-	-	-1.9
Electricity and other inputs[8]	Off-farm	12.88	16.64	14.95	-	-	-	-	-	-	-	0.8	1.8
Nitrous oxide (kg CO_2-eq/t ECM)													
Fertilizer application	On-farm	99.63	19.78	16.88	-	-	-	51.4	51.4	51.4	-1.9	34.3	-3.4
Manure storage and spreading	On-farm	34.51	82.08	153.14	-	-	-	20.1	12.8	23.4	-36.9	-15.6	-13.7
Crop residues	On-farm	2.01	6.94	3.29	-	-	-	-100.0	-20.5	-0.9	-59.2	-31.7	-40.7
Manure excreted on pasture	On-farm	139.94	4.62	0.00	-	-	-	17.7	17.7	-	-46.3	-26.0	-
Fertilizer production	Off-farm	30.83	8.72	4.73	-	-	-	-	-	-	-	-	-70.0
Concentrate production	Off-farm	7.54	36.73	52.18	-	-	-	35.4	66.4	66.2	-1.3	29.9	-45.7
Electricity and other inputs	Off-farm	6.81	8.74	8.74	-	-	-	-	2.1	5.6	-	-0.5	-8.6
Carbon dioxide (kg CO_2-eq/t ECM)													
Fuel combustion	On-farm	13.69	21.62	33.25	-	-	-	-	-	-	-	-	-
Lime application	On-farm	1.44	0.00	1.15	-	-	-	-	-	-	-	-	-
Fertilizer application	On-farm	6.71	0.00	1.61	-	-	-	-	-	-	-	-	-
Carbon sequestration	On-farm	-77.72	-17.87	0.00	-	-	-	-	-	-	-	-	-
Fertilizer production	Off-farm	43.82	11.21	9.40	-	-	-	-	-	-	-	-	-3.8
Concentrate production	Off-farm	21.44	72.24	52.70	-	-	-	-	-	-	-	-	-0.2
Land use change	Off-farm	1.81	58.02	0.00	-	-	-	-	-	-	-	-	-
Electricity	Off-farm	10.90	41.33	39.47	-	-	-	-	-	-	-	-2.5	7.7
Other inputs	Off-farm	5.19	8.37	9.07	-	-	-	-	-	-	-	-	-0.2
F-gases (kg CO_2-eq/t ECM)													
Fertilizer production	Off-farm	0.02	0.01	0.01	-	-	-	-	-	-	-	-	-
Concentrate production	Off-farm	0.02	0.07	0.04	-	-	-	-	-	-	-	-	-
On-farm, kg CO_2-eq/t ECM	On-farm	693	612	705	0.4	6.4	6.1	15.6	30.1	31.2	-12.4	-7.5	-8.7
CFP[9], kg CO_2-eq/t ECM	Total	837	877	898	0.4	4.4	4.8	13.1	23.7	28.4	-10.4	-4.2	-9.4
On-farm No Seq, kg CO_2-eq/t ECM	On-farm	771	630	705	0.4	6.2	6.1	14.0	29.2	31.2	-11.3	-7.5	-8.7
CFP No Seq, kg CO_2-eq/t ECM	Total	914	895	898	0.4	4.4	4.8	12.1	23.2	28.4	-9.4	-4.1	-9.4

[1] All GHG emissions associated with the dairy production system up to the point milk is sold from the farm expressed in kg of CO_2-equivalent per t of energy corrected milk.
[2] The baseline scenario used fixed emission factors to estimate emissions from non-agricultural inputs e.g. fuel and used the current IPCC (2006) guidelines to estimate emissions from agricultural GHG sources, except for enteric fermentation where country specific approaches were applied.
[3] S1 = Scenario 1. Fixed emission factors were used to estimate emissions from non-agricultural inputs and emission algorithms from the IPCC (2006) guidelines were applied to estimate emissions from agricultural GHG sources.
[4] S2 = Scenarios 2. Fixed emission factors were used to estimate emissions from non-agricultural inputs, and emission algorithms from the original IPCC (1997) guidelines and IPCC (2000) good practice guidelines were used to estimate emissions from agricultural GHG sources.
[5] S3 = Scenario 3. Country specific emission factors were applied to estimate emissions from the manufacture of non-agricultural inputs and from agricultural GHG sources.
[6] CO_2-eq = Carbon dioxide equivalent where CO_2 = 1, methane = 25, nitrous oxide = 298 (IPCC, 2007).
[7] ECM = Energy corrected milk = (0.25 + 0.122 × %fat + 0.077 × %protein) × kg milk (Sjaunja et al., 1990).
[8] Emissions from the production of purchased forage, milk replacer, fuel, pesticides and lime.
[9] CFP = Carbon footprint.
[10] No Seq = Carbon sequestration by permanent grassland was excluded.

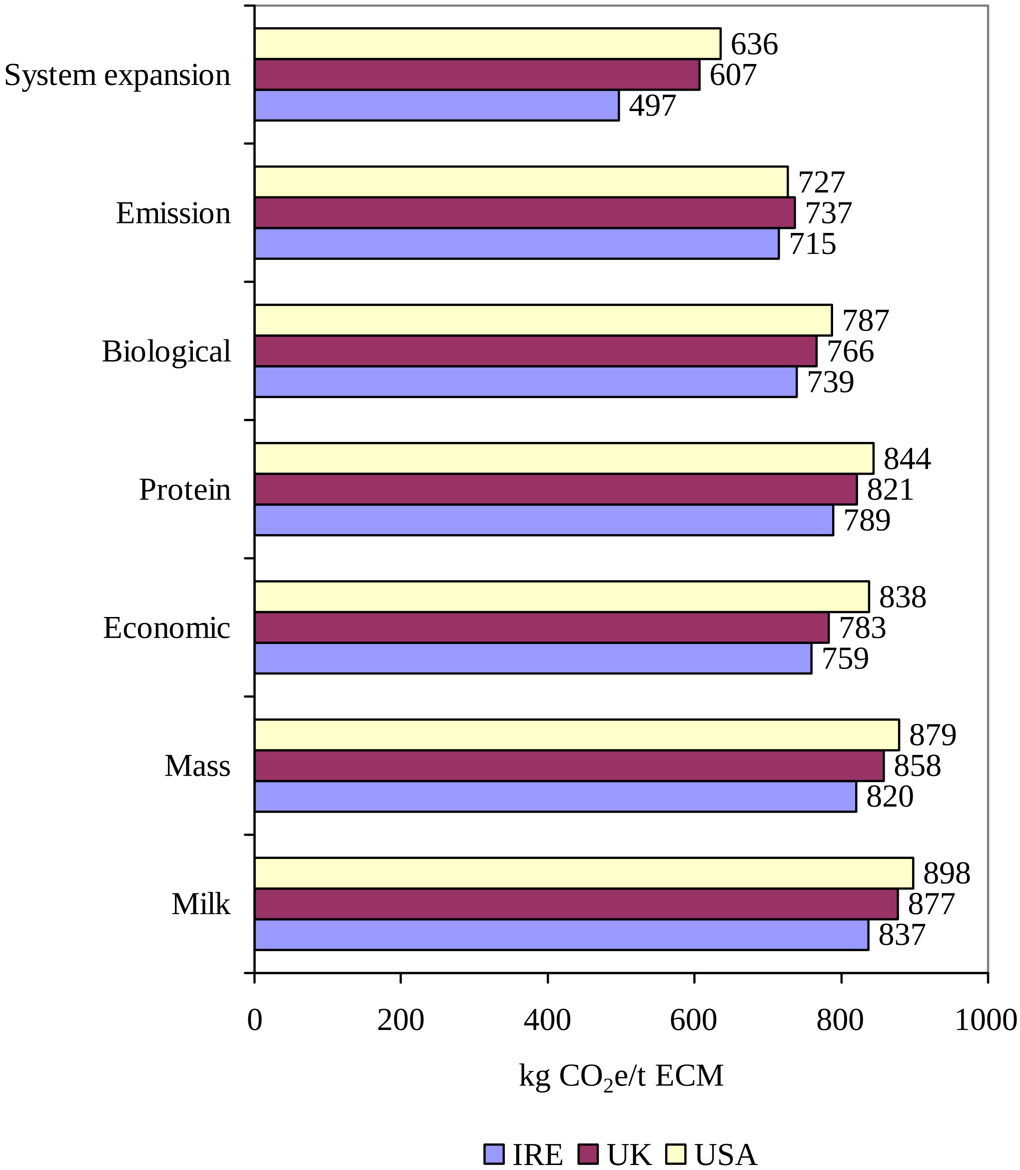

Figure 1 The effect of different methods to allocate greenhouse gas emissions between milk and meat on the carbon footprint, kg of CO_2 equivalent/t of energy corrected milk (kg CO_2e/t ECM), with carbon sequestration, of a high performance Irish grass-based dairy system (IRE), a high performance UK confinement dairy system and a top performing USA confinement dairy system. Milk = All greenhouse gas (GHG) emissions were allocated to milk. Mass = Mass of milk and meat was used to allocate greenhouse gas (GHG) emissions. Economic = Economic value of milk and meat sold was used to allocate GHG emissions. Protein = Edible protein in milk and meat was used to allocate GHG emissions. Biological = Feed energy required for producing milk and meat was used to allocate GHG emissions. Emission = The GHG emissions associated with surplus calves, dairy females <2 year of age and from finishing cows was allocated to meat with the remainder allocated to milk. System expansion = Assumes meat from milk production substitutes emissions generated by meat from traditional cow-calf beef systems.