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A review of whole farm systems models of greenhouse gas emissions from beef and dairy cattle production systems

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ABSTRACT

To comply with the United Nations Framework Convention on Climate Change (UNFCCC) greenhouse gas (GHG) emissions reporting requirements, the Intergovernmental Panel on Climate Change (IPCC) developed guidelines for calculating national GHG inventories in a consistent and standard framework. Although appropriate for national level accounting purposes, IPCC methodologies lack the farm level resolution and holistic approach required for whole farm systems analysis. Thus, whole farm systems modelling is widely used for farm level analysis. A review of 31 published whole farm modelling studies of GHG emissions from beef and dairy cattle production systems indicated a number of important outcomes. For example, improvements in animal productivity (*i.e.*, liveweight gain milk production) and fertility (*i.e.*, lower culling, lower replacement rates) can reduce GHG emissions/kg product. Additionally, intensification of production as output/ha can reduce emissions/kg product provided input requirements of feed and/or fertilizer are not excessive. Carbon sequestration into agricultural soils has the potential to offset emissions from pastoral based production systems. A product based metric is widely used and allows a wide range of objectives, including farm profitability and food security to be met. Variation in farm system parameters, and the inherent uncertainties associated with emission factors, can have substantial implications for reported agricultural emissions and thus, uncertainty or sensitivity analysis in any modelling approach is needed. Although there is considerable variation among studies in relation to quality of farm data, boundaries assumed, emission factors applied and co-product allocation approach, we suggest that whole farm systems models are an appropriate tool to develop and measure GHG mitigation strategies for livestock farms.

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Abbreviations: AFOLU, agriculture forestry and other land use; B_o, maximum CH_4 producing potential; CO_2e, CO_2 equivalents; EF, emission factor; FU, functional unit; GE, gross energy; GHG, greenhouse gas; IPCC, Intergovernmental Panel on Climate Change; LCA, life cycle assessment; SOC, soil organic C; UNEP, United Nations Environment Programme; UNFCCC, United Nations Framework Convention on Climate Change; WMO, World Meteorological Organisation; Y_m, CH_4 conversion rate.

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1. Introduction

World agriculture is currently faced with the challenge of feeding a rapidly increasing global population, predicted to peak at 9.2 billion by 2075 (FAO, 2006), while meeting an obligation to reduce greenhouse gas (GHG) emissions as climate change grows in importance as an international policy issue. Agriculture is a source of three primary GHG, being CO_2, CH_4 and N_2O. Indeed this sector has been estimated to be responsible for 10–12% of global estimated GHG emissions and ~50% of CH_4 and ~60% of N_2O from anthropogenic sources (Smith et al., 2007). The main components of agricultural emissions, outside of land use change, are N_2O released from soils related to application of N fertilizers (38%), CH_4 from livestock enteric fermentation and CH_4 and N_2O from manure management (38%), CH_4 from cultivation of rice (11%), and CH_4 and N_2O from burning of savannah, forest and agricultural residues (13%; EPA, 2006). Additionally, the agricultural sector is indirectly responsible for emissions in the industrial and energy sectors through production of inputs such as fertilizers and pesticides, production and operation of farm machinery, and on-farm energy use (West and Marland, 2002).

In general, to feed the world's increasing population in an environmentally and economically sustainable manner using current production systems and technologies, agriculture will need to place more emphasis on improved efficiencies. Although agricultural GHG fluxes are complex and heterogeneous, appropriate management of agricultural systems, such as improved agronomic practices and nutrient use, provides the opportunity for some mitigation to be achieved (Smith et al., 2007). Many available mitigation opportunities use current technologies and can be implemented immediately. Guidelines by the Intergovernmental Panel on Climate Change (IPCC) provide the best widely applicable defaults for compiling national GHG inventories. Since livestock farms, in particular, are complex systems with different interacting components including soils, crops, feeds, animals and manures, the approaches that best reduce GHG emissions will depend on local conditions necessitating specific individual approaches to evaluate appropriate mitigations.

Due to the limitations of the IPCC approaches, the concept of whole farm systems modelling was developed to describe and quantify internal cycling of materials (*e.g.*, fertilisers, animal feed, chemicals) and their constituents, as well as exchange of materials and nutrients, between the farming system and its environment. A whole farm approach is a potentially powerful instrument to predict effects of changes in management, as well as develop cost effective GHG mitigations, while ensuring that possible interactions among GHG emissions are considered. Further, production systems can also be evaluated and compared to determine best management practices.

Our main objectives were to review IPCC and whole farm approaches for modelling GHG emissions from ruminant livestock production systems with specific reference to published models, outline general conclusions from application of these published models, and describe some limitations and risks associated with these approaches.

2. IPCC reporting protocols

The IPCC was established in 1988 by the World Meteorological Organisation (WMO) and the United Nations Environment Programme (UNEP), thereby recognising the increasing evidence of anthropogenic climate change. The subsequent signature of the United Nations Framework Convention on Climate Change (UNFCCC) by ~150 countries at the United Nations Conference on Environment and Development (Earth Summit) in Rio de Janeiro in 1992 required countries to develop, update periodically and publish national GHG emissions using standard methodologies. Under terms of the subsequent Kyoto Protocol (UNFCCC, 1998), binding emissions reductions were set for 37 industrialized countries (Annex 1). In order to meet UNFCCC reporting requirements for Annex 1 countries, the IPCC published guidelines for calculating national GHG inventories (IPCC, 1997a). These were subsequently updated in 2000, 2003 and 2006 and allowed for quantification of national emissions based on readily available activity data, such as power usage, fossil fuel consumption, fertilizer sales, animal numbers and land use change, as well as associated emission factors for each activity.

Continual development of IPCC guidelines is dependent on input of global experts in the respective sectors. For example, the 2006 guidelines involved contributions from over 250 experts worldwide. The 1996 guidelines (IPCC, 1997a) required emissions to be reported in 6 categories: energy, industrial process, solvent and other product use, agriculture, land use change and forestry, and waste. The amalgamation of categories in the 2006 guidelines (IPCC, 2006) reduced the number of categories to four. Specifically, in the case of agricultural systems, agriculture and land use change and forestry were amalgamated into a single category: agriculture, forestry and other land use (AFOLU).

In terms of agriculture, the simplified approach of the IPCC protocols was predicated on wide variations in agricultural practices within and among countries, which made direct national scale measurements of farm emissions impossible. IPCC guidelines provide the best widely applicable defaults for compiling national GHG inventories and, as such, are the principle methodologies by which sectoral emissions can be compared among countries. However, the robustness of these inventories is dependent on developing country specific emission factors and verifying emissions inventories via modelling and/or direct measurement (IPCC, 1997b). Consequently, the IPCC developed a three tier system for quantifying emissions sources and sinks with each successive tier having an increased level of detail and accuracy. This allowed for increased inventory refinement where possible, while recognising that there were considerable variations in data availability, technical expertise, and inventory capacity across countries, particularly in developing countries (*i.e.*, non-Annex 1).

The IPCC disaggregates non-CO_2 GHG emissions (*i.e.*, N_2O and CH_4), into 7 categories being: enteric fermentation (CH_4), manure management (CH_4 and N_2O), rice cultivation (CH_4 and N_2O), agricultural soils (N_2O), burning of savannas (N_2O) and burning of agricultural residues (N_2O). The CO_2 sources or sinks that result in changes in biomass C or soil organic C

(SOC) are treated in the Land Use, Land Use Change and Forestry (LULUCF) sector in the 1996 guidelines (IPCC, 1997a) and in the AFOLU sector in the 2006 guidelines (IPCC, 2006). The equivalency factors for livestock systems are based on the 100 yr global warming potential for CH_4, N_2O and CO_2 of 21, 310 and 1, respectively (IPCC, 1997a). However, these equivalency factors have been updated by the IPCC (2006) and now CH_4 = 25, N_2O = 298 and CO_2 = 1.

2.1. Methane

In ruminant livestock systems, CH_4 is the predominant source of GHG emissions with enteric fermentation in the rumen comprising the majority. However, CH_4 from manure management can also comprise a substantial proportion of livestock emissions in countries where liquid manure storage systems predominate. Both sources result from microbial anaerobic respiration with CO_2 replacing O_2 as the principle electron acceptor. IPCC methodologies for generating CH_4 emission factors, be they Tier 1 or higher, are generally based on animal categories, daily feed intake values and nutritional quality of the diet. As a result, the principle uncertainties in calculating CH_4 emissions for national herd categories arise in estimating animal populations for each category and, for Tier 2, characterisation of diets and animal growth curves for each category. Indeed, the IPCC estimate a global uncertainty of ±50% for Tier 1 estimates and ±20% for Tier 2 estimates (IPCC, 2006).

2.1.1. Enteric fermentation

Enteric fermentation estimates from livestock are based on average daily feed intake as gross energy (GE; MJ/d) and CH_4 conversion rates (Y_m). In all tiers of reporting, cattle are disaggregated into two principle classes, dairy cows and others, with others further subcategorised in terms of sex, age and feeding situation. For Tier 1, default values of GE are generated from assumed values for animal body weight, average daily weight gain, diet digestibility, pregnancy status, feeding level and milk production for dairy cows for various geopolitical regions (Gibbs and Johnson, 1993). For dairy cattle and other cattle fed forage based diets, it is assumed that 6.5% ± 1% of GE intake is converted to CH_4 while, for feedlot cattle, a 3% ± 1% conversation rate is suggested. The default emission factors generated for dairy cows range from 40 kg CH_4/hd/yr for Africa/Middle East to 121 kg CH_4/hd/yr for North America. For other cattle, the regional default emission factors range from 27 kg CH_4/hd/yr for the Indian Subcontinent to 60 kg CH_4 hd/yr for Oceania and include beef cows, bulls, feedlot and young cattle (IPCC, 2006). The Tier 2 approach involves quantification of GE values generated from national and or sub-regional/provincial values of these parameters, while Tier 3 models account for more specific farm level parameters such as genotype differences, seasonal effects and variations in Y_m characteristics.

2.1.2. Manure management

The extent of GHG emissions from storage and treatment of manure depends on the amount of manure produced, its C and N content, the proportion that decomposes anaerobically, and the temperature, duration and type of storage. In general, liquid (primarily anaerobic) systems generate proportionately more CH_4, while solid systems produce more N_2O (Amon et al., 2006). For the Tier 1 approach, CH_4 emissions from cattle manure management are the product of the livestock population of any category multiplied by the emission factor (/head) for that category. The selection of this Tier 1 emission factor is based on the average annual temperature for the region and its predominant manure management system. Temperature data are based on national meteorological statistics.

The Tier 2 approach entails quantifying the quantity of volatile solids produced by livestock and the maximum amount of CH_4 (B_o) that can be produced from that manure. The portion of the maximum amount of CH_4 that can be produced from the manure is estimated using a system specific CH_4 conversion factor (MCF). The MCF varies with the manner in which manure is stored and the climate. In some cases countries directly measure B_o and MCF. When country specific values are not available, default values from IPCC (2006) are used.

2.2. Nitrous oxide

Nitrous oxide emissions occur due to nitrification of ammonium to nitrate, or incomplete denitrification of nitrate. These emissions can be direct emissions from stored manure, organic manures or inorganic fertilizers applied to soil or direct N deposition by grazing animals. In addition, indirect N_2O emissions associated with agriculture arise from volatilisation of land applied manures and/or N based fertilizers, and N lost via runoff and leaching from agricultural soils. For inventory purposes, N_2O emitted directly from stored manure is classified under the manure management subcategory and thus added to the CH_4 emissions from this category, while remaining emissions are classified under the agricultural soils category. The Tier 1 approach is predominately used to estimate N_2O emissions from agricultural soils because there is inadequate data at the local level to determine effects important variables such as soil structure and weather have on N_2O emissions from soils (McGettigan et al., 2010).

2.2.1. Manure management and pasture, paddock and range emissions

Nitrous oxide emissions from manure management systems and field urine/faecal deposition during grazing (*i.e.*, pasture, paddock, range emissions) are principally based on the amount of N excreted/hd for each population category. Total manure management emissions are the product of the amount of N excreted during storage multiplied by the associated (default or Tier 2) emission factor for that manure management system and animal population. In addition, a proportion of N that volatilises as NH_3 is considered to be re-emitted as N_2O upon wet or dry deposition to soils from N excretion by animals. Anaerobic systems tend to have lower N_2O emissions since anoxic, as opposed to anaerobic, conditions are required for partial denitrification. Pasture, paddock and range emissions are estimated from annual N excretion, less the amount of excretion during storage, for each animal population.

2.2.2. Nitrous oxide from applied fertilizers and crop residues

Direct N_2O emissions from organic and synthetic fertilizers are based on the amount of N applied multiplied by the appropriate emissions factor. Indirect N_2O emissions (N_2O_{vol}) result from volatilised N from fertilizers or industry that is subsequently wet or dry deposited or runoff and leached N from land applied or deposited N. Leached N is calculated from a proportion of total N deposited from fertilizers, animal deposition, runoff and crop residues.

2.3. Carbon dioxide

Carbon dioxide may be emitted from, or C sequestered into, agricultural soils. Net CO_2 emissions occur from agricultural practices (*e.g.*, lime application) and from net SOC release caused by land use change. Liming emissions are expressed as the amount of lime multiplied by the emission factor for limestone. The default emission factor is 0.12–0.06 t C/t limestone (IPCC, 2006).

Land use change associated with pasture land occurs if cropland, wetlands or forests are converted to grasslands. All these conversions have associated changes in total C which are comprised of changes in SOC and biomass. Measurement protocols are detailed in the good practice guide for LULUCF (IPCC, 2000). Default factors, country specific land use factors or models can be used to assess SOC change. In general, cropland conversion to pasture generally leads to SOC accumulation, while forest/wetland conversion leads to a net SOC loss (Ogle et al., 2004).

3. Whole farm modelling approaches

The structure of the IPCC reporting protocols are not conducive to integrated systems analysis as a result of the sector based approach. Specifically, emissions that arise in agricultural systems are reported in three sectors for IPCC purposes according to the 1996 guidelines (IPCC, 1997a) being agriculture, land use change and forestry, and energy. Further, indirect emissions related to agricultural production may also arise in the industrial processes and waste categories. In addition, if data from these three sectors are combined to generate a whole farm balance, any emissions generated outside of the national boundaries are not included (Cerri et al., 2009). Because of limitations of the IPCC methodology for modelling farm level emissions, which are elaborated further in Section 5, whole farm modelling is widely used (Tables 1 and 2). Whole farm GHG emissions models may be categorized as systems analysis models (Schils et al., 2005; O'Brien et al., 2010; White et al., 2010) or life cycle assessment (LCA) models (Ogino et al., 2004; Thomassen et al., 2008b; Peters et al., 2010). However, where the assumptions underpinning the analysis (*e.g.*, boundaries, functional unit, allocation method, emission factors as discussed in subsequent sections) are the same, results should also be the same. Furthermore, both methodologies follow similar approaches, although the LCA approach is somewhat more formalised (ISO, 2006b). The main phases of LCA according to the ISO standards (ISO 14040-14044) are goal and scope definition, life cycle inventory analysis, life cycle impact assessment and life cycle interpretation. For systems analysis models corresponding phases can be identified as conceptual framework definition, model development, model application and results interpretation, respectively.

3.1. Goal and scope/conceptual framework specification

In general, the first step in whole farm modelling is to define the goal and scope or, in terms of systems analysis models, to formulate a conceptual model of the farming system of interest (Fig. 1). Goal definition requires stating the aims and objectives of the study and the intended audience (ISO, 2006a). This determines if an LCA is 'attributional' or 'consequential'. An LCA study is attributional if the objective is to quantify environmental burdens from the processes and materials during the life cycle of a product at the current level of output. An LCA study is consequential if the objective is to determine changes in the total environmental burden which results from marginal change in the level of output from a product (ISO, 2006a). Attributional and consequential LCA modelling of milk production is described by Thomassen et al. (2008a).

The scope of a study involves describing the system under study and its function, and the boundaries of the studied system (ISO, 2006a). A quantitative measure of the primary product or service that a system delivers (*i.e.*, the functional unit) must be defined (ISO, 2006a). As many agricultural production systems produce more than one product, it is necessary to attribute environmental impacts to each product from the system using an allocation approach. The ISO 14040 standards (2006a) recommend avoiding allocation by splitting a multi-function process in such a way that it can be described as two separate processes, each with a single output, or, alternatively, allocation can be avoided by expanding the system boundaries of the studied system until the same functions are delivered by all systems compared. If allocation cannot be avoided, the standards advise allocation of environmental burdens between products based on the underlying physical causal relationship. When physical causal relationships alone cannot be established as the basis for allocation, then another relationship (*e.g.*, economic value of products) should be used to allocate environmental burdens among products.

Table 1
Overview of published modelling studies of greenhouse gas emissions from beef production systems.

Study	Methodology and approach used	Boundaries	Emissions factors	Result[a]
Beauchemin et al. (2010)	Multiple-year over the lifespan of a breeding animal. Based on typical Alberta, Canada crop-livestock farm finishing all cattle.	Direct on farm, purchased inputs and indirect nitrous oxide emissions. Excludes capital and machinery.	IPCC (2006) methodology for Canada.	21.7
Casey and Holden (2006a, 2006b)	Single year, whole system model. Based on typical Irish cow-calf farm finishing all cattle.	Direct on farm, purchased inputs emissions. Excludes capital, machinery and chemicals.	IPCC (1997a) for enteric fermentation and direct nitrous oxide emission. Emission factors from the literature for other values.	Casey and Holden (2006a), *20.1*; Casey and Holden (2006b), conventional, *23.2*; extensive, *21.8*; organic, *19.9*
Cederberg and Stadig (2003)	Single year, whole farm system model developed for system expansion allocation of beef from dairy system. Based on typical Swedish cow calf farm finishing all cattle.	Direct on-farm, purchased inputs and indirect nitrous oxide emissions. Excludes capital and machinery.	IPCC (1997a).	*15.6*
Crosson et al. (2010)	Single-year, whole farm system model of beef cow systems. Modelled scenarios based on typical Irish farm systems and research systems.	Direct on-farm, purchased inputs and indirect nitrous oxide emissions. Excludes capital and machinery.	Primarily IPCC (2006).	National, 21.2; Research farms, steer-beef, 19.2; Research farms, bull-beef, 18.2
Nguyen et al. (2010)	Single-year, whole farm system model developed for typical European suckler and dairy-beef production systems.	Direct on-farm, purchased inputs and indirect nitrous oxide emissions.	Primarily IPCC (2006).	27.3
Ogino et al. (2004)	Single year, whole farm system model. Based on Japanese cow-calf to finishing production system. Based on Japanese research systems.	Direct on farm, emissions from energy consumption and imported animal feed.	Empirical data from experimental systems (enteric fermentation). Literature data for other emissions.	*22.6*
Pelletier et al. (2010)	Single year, whole farm system model of cow calf, stocker and feedlot production systems. Representative of production strategies in the Upper Midwestern United States.	Direct on-farm, purchased inputs and indirect nitrous oxide emissions. Excludes capital and machinery.	IPCC (2006).	Feedlot-finished from weaning, *26.9*; Feedlot finished following store period, *29.5*; grass-finished, *34.9*
Peters et al. (2010)	An LCA model for production emissions and an input–output analysis for other emissions including purchased chemicals. Based on survey data of 3 farms (2 beef and 1 sheep) in two years (2002 and 2004).	Direct on-farm and purchased inputs emissions. Also includes emissions at the processing plant.	IPCC (1997a) methodology for Australia.	Grain-finished, 9.9; grass-finished, 12.0
Phetteplace et al. (2001)	Systems modelling of cow calf to beef American production systems. Based on average production levels for the main beef producing states.	Direct on-farm, purchased inputs and indirect nitrous oxide emissions. Excludes capital and machinery.	Primarily IPCC (1997a). Literature-sourced emission factors for enteric fermentation.	Cow calf, *37.5*; stocker, *26.2*; feedlot, *10.3*; integrated system, *28.2*
Stewart et al. (2009)	Systems modelling of four hypothetical farms representing a range of climatic and soil conditions found in Canada.	Direct on-farm and purchased inputs. Excludes capital and machinery. Includes indirect nitrous oxide emissions and land use emissions and sinks.	IPCC (2006).	In kg CO_2e/kg protein. Manitoba, 9.93; Saskatchewan, 5.65; S. Alberta, 3.28; N. Alberta, 4.64
Subak (1999)	Environmental analysis of hypothetical pastoral and feedlot beef production systems.	CH_4, and CO_2 emissions from fuel usage. Also includes emissions associated with alternative land uses.	Information not presented.	American feedlot-finished, 14.8; African pasture-finished, 8.4

Table 1 (*Continued*)

Study	Methodology and approach used	Boundaries	Emissions factors	Result[a]
Veysset et al. (2010)	Coupled a linear programming bioeconomic model with an environmental assessment model. Five demonstration farms, reflecting diverse beef farming systems, were modelled.	Direct on farm, purchased inputs and capital and machinery. Does not include indirect nitrous oxide emissions.	IPCC (2006) methodology for France.	Cow-calf, *30.5*; integrated cow-calf to beef, *26.6*
White et al. (2010)	Used farm simulation, feed formulation and nutrient budgeting models to investigate beef and sheep farming systems in New Zealand. Two representative systems were modelled; lowland and upland systems.	Direct on farm and purchased inputs. Excludes capital and machinery. Includes indirect nitrous oxide emissions for on-farm feed production but not for purchased feeds.	IPCC (1997a) methodology for New Zealand.	Lowland, ~26.0; upland, ~34
Williams et al. (2006)	Full industry approach to LCA analysis for English beef taking into account alternative production streams and the population size associated with each.	Direct on farm, purchased inputs and indirect nitrous oxide emissions. Also includes capital and machinery.	Emission factors taken from the literature.	National, 15.8; organic, 18.2; suckler, 25.3; lowland, 15.6; upland, 16.4

[a] In kg CO_2e/kg beef carcass unless otherwise stated. Results in italics indicate that these have been converted to a carcass-based output metric.

The definition of system boundaries involves specifying the parts of the product system that are included and excluded in the assessment. Generally, system boundaries of whole farm models of livestock systems are defined to assess GHG emissions from all processes up until the point the primary product is sold from the farm (Cederberg and Mattsson, 2000; Basset-Mens et al., 2009b). Emissions from production of external farm inputs (*e.g.*, concentrate feeds, fertilizers) are typically included in the analysis (Tables 1 and 2).

3.2. Life cycle inventory analysis/model development

The second step of whole farm systems modelling involves developing a mathematical model of the system defined in the initial stage. Life cycle inventory analysis involves compilation of inputs, outputs and emissions for a product system throughout its life cycle (ISO, 2006b). The aim of this stage is to develop a model which quantifies the resources used and the amount of waste and emissions generated/functional unit (Rebitzer et al., 2004). This is similar to systems analysis models where a series of equations are developed and used to mathematically model the processes and quantify outputs for the farming system. In most cases theoretical or empirical relationships are used to develop these equations (Phetteplace et al., 2001; Shalloo et al., 2004; Crosson et al., 2006; Olesen et al., 2006). Occasionally, data from the farming system under study is used (Schils et al., 2005). Equations used to estimate GHG emissions in livestock simulation models are referred to as emission factors, which are frequently obtained from IPCC guidelines to estimate emissions produced on farm (IPCC, 1997a, 2006). Because IPCC emission factors are designed to enumerate national level emissions, they may lack the refinement to quantify effects that changes to the production systems have on GHG emissions on individual farms (Schils et al., 2006). Thus, representative emission factors that have recently been published in the scientific literature should be used. International databases (BUWAL, 1996; Ecoinvent Centre, 2009) or literature sources are used to estimate resources used and emissions generated from processes that are indirectly related to the production system of interest (*e.g.*, data on fuel and fertilizer production).

3.3. Life cycle impact assessment/model application

Once the quantitative model is developed, the next step is to apply the model to analyse the farming system under study. For beef systems, several models have been developed for pasture based and confinement or feedlot systems (Subak, 1999; Phetteplace et al., 2001; White et al., 2010). For dairy production systems, the majority of models have been developed to quantify GHG emissions from pasture based systems (Lovett et al., 2008; Beukes et al., 2010; O'Brien et al., 2010). Relatively few models have been developed to estimate emissions from confinement systems (Chianese et al., 2009; Rotz et al., 2010). Usually, data collected on farm or representative farm information such as regional statistics are used as input data to operate these quantitative GHG models. In addition to GHG emissions, LCA models often have additional impact categories including land use, energy use, acidification potential and eutrophication potential (Cederberg and Stadig, 2003; Ogino et al., 2004; Williams et al., 2006; Nguyen et al., 2010).

Table 2
Overview of published modelling studies of greenhouse gas emissions from dairy production systems.

Study	Methodology and approach used	Boundaries	Emissions factors	Result[a]
Basset-Mens et al. (2009b)	LCA of average New Zealand dairy farming systems (based on national statistics) compared to alternative scenarios varying in intensity.	Direct on farm, purchased inputs and indirect nitrous oxide emissions.	IPCC (2006) methodology for New Zealand.	National, 0.933; low input, 0.646; high N, 0.762; intensive maize, 0.754
Beukes et al. (2010)	Integrated three models (a dynamic, mechanistic whole farm simulation model, a mechanistic animal model and nutrient flow model) to investigate mitigation scenarios for typical New Zealand dairy systems.	Direct on farm, purchased inputs and indirect nitrous oxide emissions.	IPCC methodology for New Zealand.	0.912
Casey and Holden (2005a,b)	Single year, whole farm system model. Based on typical Irish milk production systems.	Direct on farm, purchased inputs emissions. Excludes capital, machinery and chemicals.	IPCC (1997a) for enteric fermentation and direct nitrous oxide emission. Emission factors from the literature for other values.	Casey and Holden (2005a), 1.46; Casey and Holden (2005b), conventional, 1.08; extensive, 1.2
Cederberg and Stadig (2003)	Single year, whole farm system model developed to investigate allocation options for dairy systems. Based on typical Swedish dairy farms.	Direct on farm, purchased inputs and indirect nitrous oxide emissions. Excludes capital and machinery.	IPCC (1997a).	1.05
Gerber et al. (2010)	LCA of the global dairy industry disaggregated based on region and production system.	Pre-farm gate (direct on farm, purchased inputs and indirect nitrous oxide emissions) and post-farm gate (from the farm gate to the retail point).	IPCC (2006).	1.5, Western Europe; 1.1, North America
Haas et al. (2001)	LCA using real-farm data from six conventional, six extensive and six organic German dairy farms.	Insufficient information to clarify precise boundaries used.	Emission factors taken from the literature.	Intensive, 1.3; extensive, 1.0; organic, 1.3
Lovett et al. (2006, 2008)	Coupled a bioeconomic simulation model to a whole farm system GHG model to investigate a number of research farm systems differing in cow genetic merit and feeding system.	Direct on farm, purchased inputs and indirect nitrous oxide emissions. Excludes capital and machinery.	Emission factors taken from a range of literature sources.	Lovett et al. (2006), 1.04–1.15; Lovett et al. (2008), free-draining soils, 0.89; heavy soils, 0.99
O'Brien et al. (2010)	Coupled a bioeconomic simulation model and a whole farm system GHG model to investigate a number of research farm systems differing in cow genetic merit and feeding system.	Direct on farm, purchased inputs and indirect nitrous oxide emissions. Excludes capital and machinery.	Emission factors taken from a range of literature sources.	0.73–0.81
Olesen et al. (2006)	A dynamic simulation model of C and N flows on dairy farms to assess mitigation measures and strategies. Modelled five hypothetical European dairy farms in different zones.	Direct on farm, purchased inputs and indirect nitrous oxide emissions. Excludes capital and machinery.	Three sets of emission factors used: IPCC (1997a) Tier 1, IPCC (2000) Tier 2 and default model values.	Conventional, 1.43; organic, 1.57
Phetteplace et al. (2001)	Systems modelling of American dairy production systems. Based on average production levels for the main dairy producing states.	Direct on farm, purchased inputs and indirect nitrous oxide emissions. Excludes capital and machinery.	IPCC (1997a) for all except enteric fermentation where literature-sourced emission factors were used.	1.09

Table 2 (*Continued*)

Study	Methodology and approach used	Boundaries	Emissions factors	Result[a]
Rotz et al. (2010)	Whole farm system, semi-mechanistic GHG simulation model. Modelled pasture-based and confinement US dairy production systems.	Direct on farm, purchased inputs and indirect nitrous oxide emissions. Excludes buildings.	Primarily IPCC (2006) and literature-sources emission factors for secondary emission sources.	Pasture-based, 0.62; confinement, 0.46–0.69
Schils et al. (2005)	Static whole farm system model of the annual flows and emissions of C and N in dairy systems. Two dairy systems on the same site but differing in intensity were modelled.	Direct on farm, purchased inputs and indirect nitrous oxide emissions. Also includes carbon sequestration.	Emission factors taken from a range of literature sources.	Conventional, 0.7; grass-clover, 0.63
Thomassen et al. (2008b)	Cradle to farm gate analysis of ten conventional and 11 organic Dutch dairy production systems.	Direct on farm, purchased inputs and indirect nitrous oxide emissions. Excludes chemicals and buildings.	Emission factors taken from the literature for most sources (IPCC (1997a) emission factors for nitrous oxide from managed soils).	Conventional, 1.4; organic, 1.5
Williams et al. (2006)	Full industry approach to LCA analysis for English dairy production taking into account alternative production streams and the population size associated with each.	Direct on -farm, purchased inputs and indirect nitrous oxide emissions. Also includes capital and machinery.	Emission factors taken from the literature.	Conventional, 1.03; organic, 1.19; high maize, 0.95; high production, 0.99; split calving, 1.00

[a] In kg CO_2e/kg milk.

3.4. Interpretation of results

Following model application, results are assessed and conclusions and recommendations are formulated to improve the production system. Data from previous livestock simulation GHG models have been quantified/farm, /ha⁻of farmland and /kg of product (*e.g.*, /kg of beef or /kg of milk; Phetteplace et al., 2001; Lovett et al., 2006; Basset-Mens et al., 2009b). The various approaches used to express emissions are, in many cases, dependant on the objective of the study. When results are quantified on a /kg product basis some studies allocate GHG emissions among products produced on farm (*e.g.*, a dairy farm producing both meat and milk; Cederberg and Stadig, 2003; O'Brien et al., 2010) while others use no allocation (Schils et al., 2005; Beukes et al., 2010). The data resulting from whole farm systems GHG emissions models are normally tested using sensitivity or uncertainty analysis (Olesen et al., 2006), and assessed to determine which sources contribute most to emissions from the farming system, as well as the potential effect that various modifications to the production system may have on emissions.

4. Applications

An overview of recently published studies of GHG emissions from beef and dairy production systems is in Tables 1 and 2. There are 15 modelling studies for beef production systems (Table 1), which do not include studies describing emissions from dairy calf to beef production systems which are generally much lower than those in beef cow systems as a result of cow GHG emissions being mostly allocated to milk production for dairy systems (Williams et al., 2006; Verge et al., 2008). There were 16 dairy production studies reviewed (Table 2).

4.1. Modelling approaches

In many cases, the approach used was to integrate a number of models which were then utilised to complete various components of the analysis required. For example, White et al. (2010) used a farm simulation model, a feed formulation program and a nutrient budgeting model simultaneously to investigate New Zealand cattle farming systems. Similarly, Crosson et al. (2010), Peters et al. (2010) and Veysset et al. (2010) used a multiple model approach to quantify GHG emissions from beef production systems.

For dairy systems, 5 studies coupled two or more models for different aspects of the analysis (Lovett et al., 2006, 2008; Beukes et al., 2010; O'Brien et al., 2010; Rotz et al., 2010). Olesen et al. (2006) applied a whole farm GHG model to investigate European dairy farm systems in various regions, while Schils et al. (2005) developed a farm level accounting method to model Dutch dairy research farm systems.

Most models were single year whole farm system studies with the exception of Beauchemin et al. (2010), who adopted a multi year approach using the lifespan of a breeding female as the time reference, and Williams et al. (2006) and Gerber

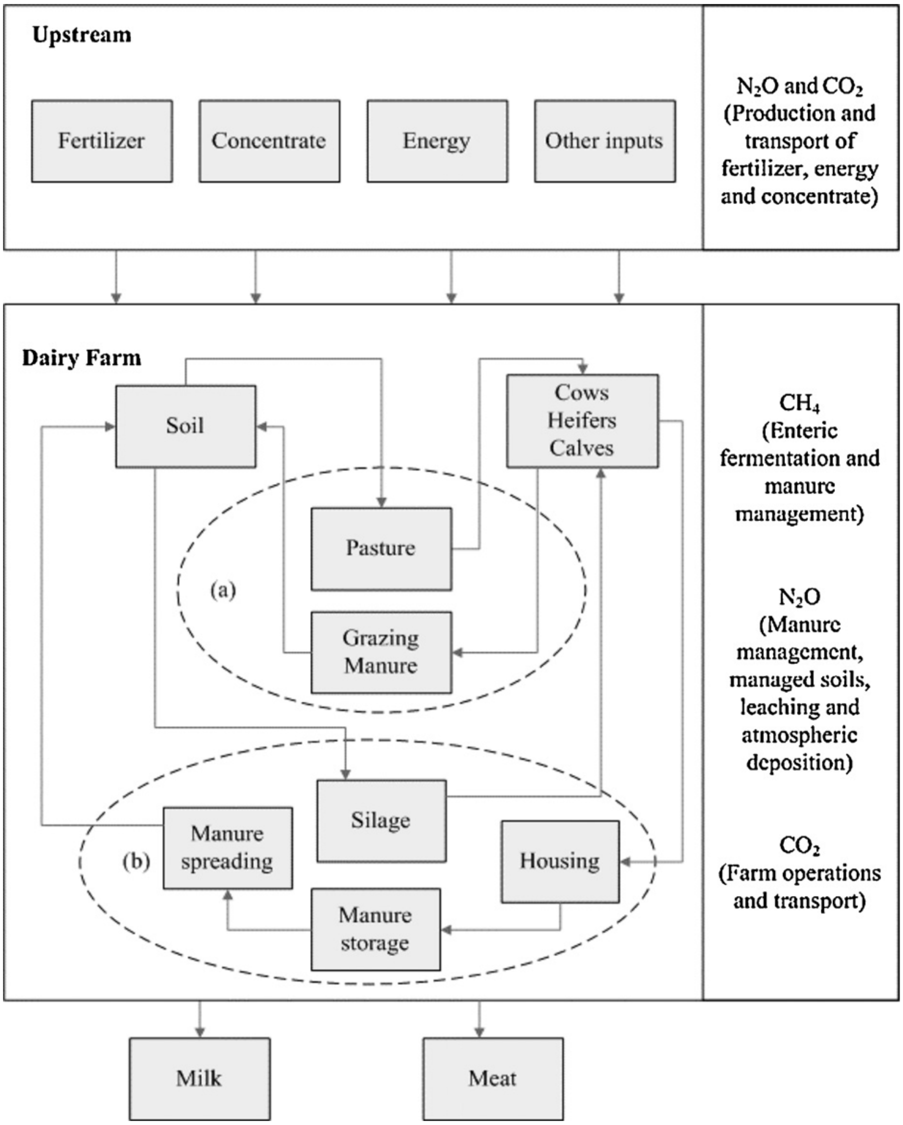

Fig. 1. Representative flow chart of a pastoral dairy farming system and associated GHG emissions: N_2O, CO_2 and CH_4. The letters (a) and (b) represent the grazing and housing feed subsystems, respectively (from O'Brien et al., 2010).

et al. (2010) who adopted industry approaches to national and global emissions, respectively. While industry and global approaches are sufficient for accounting emissions, they are less suitable for investigating impacts of specific technological interventions or improvements such as use of a particular abatement strategy. As such, the analysis of Williams et al. (2006) is more likely to yield a representative national value for GHG emissions from English beef production systems given the industry level farm data used. In contrast, the approach of Beauchemin et al. (2010), and the other studies in Tables 1 and 2, can facilitate investigation of implications of alternative production strategies on GHG emissions for the farming systems modelled. Furthermore, the analysis of Gerber et al. (2010), whereby the global dairy industry is divided into wide logical regions which could be expected to have substantial among farm variation within region may yield an accurate picture of global and regional level emissions/kg product, but is less suited to more detailed farm level analysis.

4.2. Analysis of production systems

The analysis completed using these modelling approaches is typically an accounting of GHG emissions from a number of representative farming systems. There is little attempt to address mitigation strategies, particularly in the beef production studies with the exception of Stewart et al. (2009). Rather, the effect of moving from one production system to another is analysed with regard to its effect on resultant GHG emissions. It is clear from the results that there is a substantial range in GHG emission/kg product (Table 1). Mitigation is addressed more specifically in the dairy systems analysis (Schils et al.,

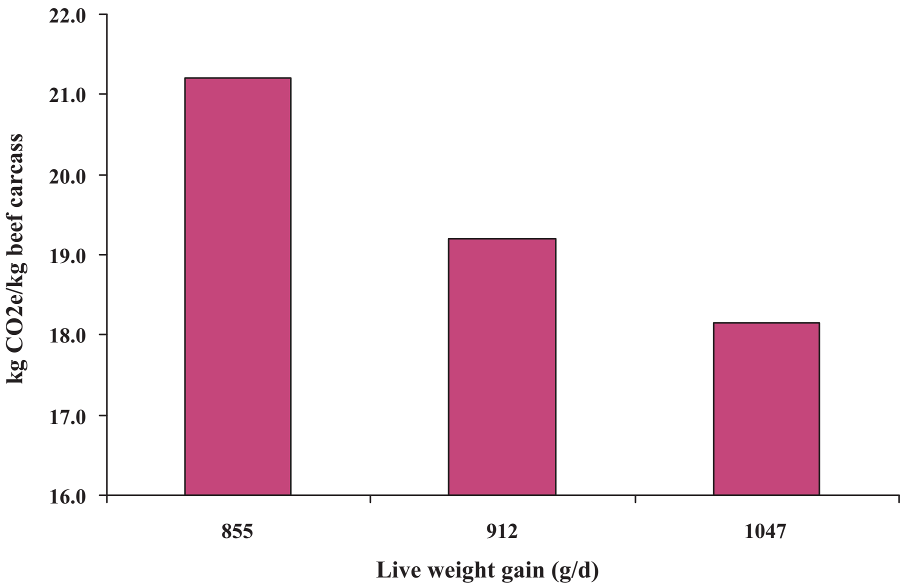

Fig. 2. Implications of level of animal performance (liveweight gain; g/d) on GHG emissions for Irish suckler beef production systems (from Crosson et al., 2010).

2005; Weiske et al., 2006; Lovett et al., 2008; Stewart et al., 2009; Beukes et al., 2010). Nevertheless, most authors focus on accounting for system level effects on emissions. Similar to the beef systems analyses, the range in results is large (Table 2).

Much of the difference among modelling studies reflects differences in modelling methodologies used, and in emission factors applied. However, these differences can also be partly explained by inherent differences among the production systems investigated. Nevertheless, it is possible to draw some general conclusions from these analyses.

4.2.1. *Animal performance*

Improvements in liveweight gain for Irish beef production systems was found by Casey and Holden (2006b) and Crosson et al. (2010; Fig. 2) to be an important mitigation strategy with, for example, beef production systems based on use of finishing bulls compared to steers reducing emissions by 6 and 5%/kg beef, respectively. A further 6 studies compared grain fed feedlot systems, with high levels of animal performance, with grassland pastoral systems. In 2 of these studies (Subak, 1999; Peters et al., 2010) integrated systems from calf to beef were modelled with Peters et al. (2010) finding that grain fed systems had lower emissions while Subak (1999) found the opposite. However, the systems compared in Subak were with different regions, breed types and, presumably, levels of management and technical efficiency making it is difficult to draw definitive conclusions. The remaining 4 grain fed *versus* grass fed studies (Phetteplace et al., 2001; Beauchemin et al., 2010; Pelletier et al., 2010; Veysset et al., 2010) were conducted in the context of the 3 components of cow calf to beef production systems being, cow calf, stocker (*i.e.*, the period between weaning and start of finishing), and the feedlot. In all cases, the cow calf phase had the highest emissions/kg product largely due to the relatively higher emissions from beef cows compared with younger non-lactating animals. In these studies, the feedlot phase had the lowest emissions with the stocker phase being intermediate.

Variations in animal performance levels on GHG emissions from dairy production systems is somewhat less clear, although improved milk performance/cow can reduce emissions (Casey and Holden, 2005a,b; Schils et al., 2005; Beukes et al., 2010; Rotz et al., 2010). However, if a breeding strategy aimed at improving lactational performance resulted in impaired fertility and, consequently longer calving intervals and higher culling rates, overall emissions may increase (Lovett et al., 2006; Beukes et al., 2010; O'Brien et al., 2010). This is supported by Weiske et al. (2006), who found that a reduction of 10% in replacement rate, combined with a strategy to sell surplus heifers at birth, reduced total emissions by 10%. It is apparent that a balanced breeding strategy optimising milk production capacity while minimising the number of non-milk producing cattle is important with regard to minimising emissions from dairy production systems.

4.2.2. *Intensification of production*

Intensification, defined as increased output/ha, invariably led to increased emissions when expressed on an area basis. However when expressed on a product basis, the result was less obvious. Improvements in animal performance were coupled with increasing intensity of production in the analysis of Crosson et al. (2010), where average farming conditions from national survey data and research farm conditions were modelled for Irish beef production systems. In that study, when increasing stocking intensity (+1 livestock unit (LU)/ha) was allied to higher levels of animal performance (+60 g/d over the lifetime of finishing cattle) recorded under research farm conditions, total emissions were reduced by 17%/kg beef carcass.

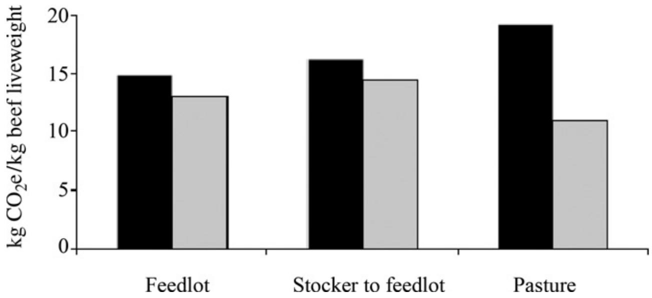

Fig. 3. Greenhouse gas emissions/kg beef liveweight for feedlot and grass based finishing systems in the United States assuming either equilibrium conditions for soil organic C (grey bars) or 0.12 kg C sequestered/ha/yr for cow calf systems and 0.4 t C sequestered/ha/yr for intensive grazing (black bars; from Pelletier et al., 2010).

However, a further incremental increase in stocking rate (+0.7 LU/ha) increased GHG emissions by 11%/kg beef carcass, largely due to the substantial increase in N fertilizer application required at high stocking rates. These findings were mirrored by Casey and Holden (2006a) and White et al. (2010) who found that increasing N application increased GHG emissions/kg beef. In the analysis of White et al. (2010), alternative approaches to increasing intensity by feeding maize silage or finishing cattle in feedlots, as opposed to pasture based finishing, reduced emissions/kg beef, particularly compared to increasing N fertilizer usage.

Impacts of intensification of dairy production systems were also investigated by a number of authors, in many cases through comparison of organic and conventional production regimes. For example, in modelling Dutch dairy systems, Thomassen et al. (2008b) found that conventional production systems had lower emissions/kg milk than organic production systems. In contrast, Haas et al. (2001) found no difference between organic and conventional intensive production systems. However, this latter study also found that conventional extensive production systems had lower emissions/kg milk than conventional intensive or organic production systems. This is supported by Basset-Mens et al. (2009b), who reported that increased production intensity in New Zealand production systems, in terms of output/ha, increased emissions/kg product. Casey and Holden (2005b) found no relationship between stocking rate and GHG emissions/kg milk for 10 Irish dairy farms. The apparent inconsistency in results may be due to the form of intensification and the corresponding increase in productivity. Haas et al. (2001) and Basset-Mens et al. (2009b) investigated increased production/ha through higher N application rates. In both cases, the increase in N applied was relatively large compared to the increase in productivity with, on average, N application increasing by more than 100 kg/ha for a 23% increase in milk production. Basset-Mens et al. (2009b) also looked at increasing production intensity by incorporating purchased feed into the farm system. In this case, although productivity increased substantially, GHG emissions associated with the supplementation also increased substantially. The overall impact was that total GHG emissions/kg milk were higher than in the extensive scenario.

Where increasing intensity in terms of output/ha led to reduced emissions (Thomassen et al., 2008b), productivity increases were much more than those of Basset-Mens et al. (2009b) or Haas et al. (2001), at 65%. Thus, although N application rates also increased, the overall effect was to reduce total GHG emissions.

Overall, results indicate that increased output/ha through increased intensification can reduce emissions/kg product, provided that excessively high levels of N fertilizer use can be avoided and that overall emissions associated with intensification are offset by higher levels of productivity. A review by Burney et al. (2010) further demonstrates the importance of agricultural intensification, and suggests that land use changes avoided due to agricultural intensification have more than offset emissions resulting from intensification from 1961 to 2005.

4.2.3. Carbon sequestration

In most studies above it was assumed that the soil C balance was in equilibrium and modelled simulations of C sequestration have demonstrated that grasslands may reach this equilibrium after 20–40 yr depending on soil type (Hutchinson et al., 2007). However, several authors (Conant et al., 2001; Soussana et al., 2004; Byrne et al., 2005; Jacksic et al., 2006) have suggested that, as well as being large C sinks, permanent grassland soils can have an important role in sequestering C, particularly where improved grazing strategies have been adopted. Veysset et al. (2010), citing Hacala et al. (2006), noted that the ‘C offset’ when sequestration is taken into consideration could be as high as 40–70% of total GHG emissions from grassland based systems.

Pelletier et al. (2010) investigated aspects of pastoral beef production and found that, if literature estimates of C sequestration rates for conditions similar to that for the regions and systems modelled in that study were assumed, emissions were reduced by 43%. Additionally, making allowance for sequestration potential, the cow calf pastoral component of production had the lowest GHG emissions, rather than the highest (Pelletier et al., 2010; Fig. 3). Similarly, Schils et al. (2005), in comparing grass and grass/clover systems for dairy production, estimated C sequestration rates at 6.5 t CO_2e/ha for grass systems and 5 t CO_2e/ha for grass/clover systems. Total dairy emissions were reduced by 37%/kg milk in this latter case. Rotz et al. (2010) found that a GHG offset of 10–22% occurred where a confinement based dairy system was converted to a pasture based system, although they pointed out that the magnitude of this offset is likely to diminish over a period of 50 yr.

Subak (1999) also looked at C storage for African Sahelian and American feedlot beef production systems and assumed that Sahelian pastures had a lower 'opportunity cost C uptake potential' than cultivatable land. This analysis showed that total GHG emissions were increased by 63 and 22%/kg beef for American feedlot and African pastoralist systems, respectively. Nguyen et al. (2010) adopted a similar approach and specified impacts of land use and land use change in terms of the opportunity cost of potential sequestration if land reverted to forestry. Additionally, this study took into account the potential consequent land use change as a result of animal feed requirements. Results indicated that full consideration of these land use aspects could increase emissions/kg beef threefold compared to a baseline where no land use effects are considered.

However, despite its obvious importance to overall accurate accounting procedures, difficulties still remain regarding the C sequestration potential of soils due to temporal and spatial uncertainties (Gottschalk et al., 2007).

4.2.4. Choice of functional unit

The choice of functional unit of GHG emissions has important implications for interpretation of results, particularly where intensification of production is investigated. All studies describe emissions on a /kg product basis, typically /kg liveweight or carcass weight for beef production systems and /kg of milk, often corrected for solids content for dairy production systems. The two exceptions to this are studies of Cederberg and Stadig (2003) and Stewart et al. (2009) who expressed output as bone-free meat and protein, respectively. Many studies also provide an area based output measure, either /ha (Haas et al., 2001; Casey and Holden, 2005b; Beukes et al., 2010; White et al., 2010) or /farm (Casey and Holden, 2005a,b; Lovett et al., 2006; Pelletier et al., 2010).

White et al. (2010) showed that, although GHG emissions/ha increased with increasing intensification (*i.e.*, by feeding maize silage), when productivity increases were considered, emissions/kg beef decreased. This is consistent with Crosson et al. (2010) who showed that, although intensification increased emissions/farm and /ha, emissions/kg beef decreased if N fertilizer application rates did not become excessively high. The analysis of Stewart et al. (2009) corroborates this finding in that strategies which reduce productivity (*e.g.*, halving N application to grazing land, eliminating fertilizers from forage production) reduce total farm GHG emissions but increase emissions/kg protein produced. The exception to these findings is Casey and Holden (2006a) who showed the opposite effect with both emissions/ha and /kg beef increasing with increasing intensification for Irish beef farms. However, in their study, the increase in stocking rate was due to two factors, increasing N application and increased concentrate feeding, and the corresponding increase in productivity was relatively modest (+15%). In effect, although stocking rate and productivity increased, efficiency of production decreased.

4.2.5. Variation and uncertainty in GHG emissions modelling

The variation surrounding farm system input and output parameters, and inherent uncertainties with emission factors (*e.g.*, N_2O emissions from N fertilizer application) can have substantial implications for reported agricultural emissions (see Basset-Mens et al. (2009a) for more detailed discussion). Haas et al. (2001), Phetteplace et al. (2001) and Thomassen et al. (2008b) take among farm variation into account using regional livestock statistic data or survey data, and results indicate that impacts of variation in farming systems can be substantial with regard to the range in GHG emissions calculated. In modelling German dairy farms, Haas et al. (2001) showed that maximum emissions calculated were 55, 33 and 17% of minimum potential emissions for intensive, extensive and organic systems, respectively. Thomassen et al. (2008b) represented the range in emissions due to variations in farm system parameters for Dutch dairy farms by its standard deviation, and found a coefficient of variation of 7 and 20% for conventional and organic farms, respectively. Similarly, Phetteplace et al. (2001) found a coefficient of variation of 15% for American cow calf to beef production systems. It is clear that use of representative farm data is necessary to ensure that results are a valid representation of the modelled farm system or region.

A second area of uncertainty concerns emission factors, as spatial, temporal and weather induced uncertainty can reduce the robustness of emission factors and, thus, Gibbons et al. (2006) suggested that explicit consideration of uncertainty is very important, particularly when modelling farm adaptation. A number of studies consider uncertainty in emission factors (Casey and Holden, 2005a,b, 2006a,b; Gibbons et al., 2006; Lovett et al., 2006, 2008; Basset-Mens et al., 2009a). Two approaches were used. Casey and Holden (2005a,b, 2006a,b) and Lovett et al. (2006) used different fixed emission factors for the most important emission sources (*e.g.*, enteric fermentation) and presented a range of outputs based on these factors. This can be considered a form of sensitivity analysis rather than explicit uncertainty analysis. The second approach utilised stochastic modelling such as Monte Carlo simulation (Gibbons et al., 2006; Lovett et al., 2008; Basset-Mens et al., 2009a), which requires development of probability distributions of uncertain model parameters. The selection of appropriate probability distributions based on published values, and subjective assessment of the most relevant emission factors for the production system and set of farm conditions modelled is important (Lien, 2003). Random draws are taken from these distributions and the process is repeated a number of times resulting in an output distribution based on those random draws. Results are typically presented as frequency distributions (Gibbons et al., 2006) or cumulative density functions (Lovett et al., 2008; Basset-Mens et al., 2009a) where outcomes indicate the range in GHG emissions for any production system. This procedure can also be used to establish stochastic dominance of one system over another. Stochastic dominance can be used to separate risky scenarios and identify efficient options for decision-makers (Olynk and Wolf, 2008). Lovett et al. (2008; Fig. 4) showed that dairy farming on a site with 250 grazing d/yr was stochastically dominant over a site with 149 grazing d/yr with regard to GHG emission (*i.e.*, emissions were lower at any level of probability). Although many studies do not consider uncertainty in emission factors, a number of authors use emission factors that are considered more representative of the region or farm system under investigation (*e.g.*, Schils et al., 2005) and, therefore, the uncertainty surrounding these are less than where

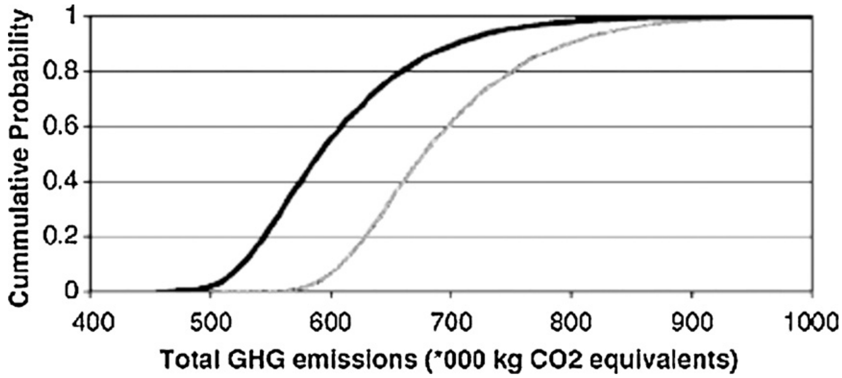

Fig. 4. Cumulative probability distribution showing the influence of key input variables on GHG emissions from Irish pastoral dairy production systems on a site with a long grazing season (Moorepark (black line); 250 grazing d/yr) and a site with a short grazing season (Kilmaley (grey line); 149 grazing d/yr) (from Lovett et al., 2008).

global or regional defaults are used. For example, Stewart et al. (2009) includes site specific emission factors that permit implications of alternative site specific factors such as soil type to be ascertained.

4.2.6. Modelling of mitigation strategies

A number of modelling studies explicitly address mitigation options for beef (Stewart et al., 2009) and dairy production systems (Schils et al., 2005; Weiske et al., 2006; Lovett et al., 2008; Beukes et al., 2010; Rotz et al., 2010). A critical aspect of these models is that they be sensitive to local conditions and/or variations in production practices. As such, default Tier 1 IPCC emission factors are not sufficiently sensitive to capture many of the production scenarios modelled. Weiske et al. (2006), Lovett et al. (2008), Stewart et al. (2009), and Rotz et al. (2010) applied whole farm models to investigate mitigation strategies for Europe, Ireland, Canada and the USA, respectively. Results highlighted the importance of the sensitivity of these models to local conditions. For example, Stewart et al. (2009) showed that elimination of fertilisation for forages had a modest impact on emissions for beef farmers in Alberta (Canada) but resulted in large increases in emissions/kg product for farmers in Saskatchewan and Manitoba (Canada) due to differing yield potentials among locations.

For dairy farming in Ireland, Lovett et al. (2008) showed that development of whole farm mitigation plans are site specific, and recommended that calving date be dependent on date of turnout from housing to pasture in spring for spring calving cows which, in turn, depends on local soil and climatic conditions. For similar grass based dairy production systems in New Zealand, Beukes et al. (2010) investigated mitigation of GHG emissions by removing cows from pastures during wet conditions and using nitrification inhibitors to reduce nitrification and N_2O emissions. In this case only one site was assumed, with results showing that removing cows resulted in no reduction in emissions, but that nitrification inhibitors reduced emissions modestly. The potential of biogas production by anaerobic digestion for European and American dairy farms was modelled by Weiske et al. (2006) and Rotz et al. (2010). Weiske et al. (2006) found impacts to be highly variable (*i.e.*, total farm emissions ranging from +24% to −96% of baseline) and largely dependent on substitution of fossil fuels with biogas. In contrast, Rotz et al. (2010) found that biogas production from enclosed manure storage systems reduced emissions by 39%/kg milk. Weiske et al. (2006) found more consistent impacts with improved manure management, but reductions in GHG emissions were modest.

Limitations of IPCC approaches are highlighted by farm level methodologies adopted in the studies. For example, Schils et al. (2005) noted that the IPCC methodology lacks the refinement necessary for mitigation analysis. Futhermore, Weiske et al. (2006) noted that farm level modelling is critical since mitigation measures may have differing abatement potentials at the farm and at the farm component level. Thus, farm level evaluations are necessary to support policy makers with regard to development of GHG emissions mitigation strategies. In the absence of farm level evaluation, mitigation strategies may have lower than expected abatement outcomes and unintended adverse consequences.

5. Critique of IPCC and whole farm systems methodologies

We have briefly reviewed IPCC guidelines for reporting national GHG emissions. We have also described approaches to modelling whole farm GHG emissions and summarised main outcomes from a number of published studies where these approaches were applied to quantify emissions and mitigation options for beef and dairy cattle production systems. The next section addresses the appropriateness, limitations and issues associated with these modelling approaches in the context of farm level GHG emissions modelling.

5.1. IPCC methodology

Guidelines developed and published by IPCC will continue to be the primary methodology for reporting national emissions. Schils et al. (2007) noted that whole farm modelling approaches should not be seen as a replacement for the IPCC methodology. This is also supported by a synthesis of the results we have presented, whereby variation in methodology, emission factors used and production systems modelled preclude rigorous comparison of results among the various studies. IPCC guidelines overcome these issues by providing a consistent and transparent approach of accounting national emissions. Nevertheless, there are a number of shortcomings with the IPCC methodology with regard to modelling whole farm systems.

The objective of the IPCC guidelines is to model national level emissions and, as such, variations that arise due to differences in farming systems and regions are not considered. Tier 1 emission factors are often used which, although suitable at macro-scale, are less suited at micro-scale (Pelletier et al., 2010). Together these factors serve to mask the considerable variability which occurs among farms. Additionally, since the IPCC approach reports emissions on a national basis, international trading of products can inadvertently lead to incorrect conclusions with regard to emissions from the agricultural sector. For example, national animal feed consumption is typically comprised of feed produced within a country together with imported feedstuffs. In this case, farming systems requiring high levels of imported supplementary feeding may appear to be more efficient from a GHG emissions perspective than a corresponding system utilising high levels of feed produced in the country. This could lead to incorrect conclusions with regard to the ranking of systems of agricultural production in terms of GHG emissions efficiency.

The structures of the IPCC methodology, whereby emissions are reported in 6 sectors using the 1996 guidelines (IPCC, 1997a), are not consistent with the integrated nature of agricultural production systems. More specifically, GHG emissions that emanate from livestock farms are reported in the following sectors for countries using the 1996 guidelines (IPCC, 1997a). These are energy (*e.g.*, electricity), industrial processes (*e.g.*, production of chemical fertilizers), agriculture and land use change, and forestry. Therefore, where farm emissions are calculated based exclusively on the quantity reported in the agriculture sector of the reporting protocol, total farm emissions are underreported with the degree being dependent on the nature of the system under investigation. This may have important implications for policy development. For example, fossil fuel energy intensive farming systems or farming system developments requiring land use changes may not be adequately addressed. This may also be accentuated by imported products, which are not reported in national emissions as noted previously.

Under the terms of the Kyoto Protocol, the metric used to measure GHG emissions is total emissions/nation state, with differences in economic status or per capita emissions recognised only in terms of assigned Kyoto targets. As a result, productivity of agricultural systems as GHG emissions/kg livestock product is not considered. Given the substantial increase in global demand for livestock products (FAO, 2006), the issue of food security is of increasing importance. This issue is beyond the scope of the UNFCCC agreements, and the consequent IPCC reporting guidelines, which is primarily concerned with emissions of GHG in regions and/or countries. The potential consequences of this are twofold. Firstly, addressing the issue of GHG emissions in isolation from food production could exacerbate future food security. The food crisis of 2008, which affected large parts of the developing world, (FAO, 2009) illustrated the precarious nature of global food supply and demand and, thus, both GHG emissions and food security issues must be considered concurrently. Secondly, given that demand for agricultural products is increasing (FAO, 2010), any contraction in food production in one region or country to meet GHG emissions reduction targets will likely result in increased production in another. This issue, termed ‘C leakage’, could potentially result in a global increase in GHG emissions if the region where production increases as a result of contraction elsewhere had a higher GHG quotient (*i.e.*, GHG emissions/kg livestock product) than the region where production had contracted. This could have potentially substantive impacts on global GHG emissions.

5.2. Whole farm modelling approaches

To overcome limitations associated with IPCC methodologies, whole farm systems modelling approaches are widely used. This enables all emissions associated with production of livestock products to be calculated and, typically, this involves a cradle-to-farm gate approach. There are a number of components of whole farm systems models that require specific attention with respect to comparing results, as well as future model development.

5.2.1. Data sources

The first issue involves the source of farm level data. This was identified in section 4.2.5 in the context of implications of farm level data variation on reported GHG emissions. In the case of Veysset et al. (2010), real farm data were taken from five ‘test cases’, or demonstration farms, which were chosen for analysis based on the degree that they represented typical French beef farming situations, whereas Schils et al. (2005) used real data from two Dutch research farms. Lovett et al. (2006, 2008) and O’Brien et al. (2010) also used research farm data, in this case data was used to parameterise whole farm systems models to specify input parameters for GHG models. Six of the beef studies referred to earlier used representative farm data such as national statistics. Two studies used actual farm data based on survey data and questionnaires (Casey and Holden, 2006b; Peters et al., 2010). Five of the dairy studies used modelled representative farms for the region and production systems under investigation, two studies used actual farm data (Haas et al., 2001; Casey and Holden, 2005b) and one study used a combination of both (Basset-Mens et al., 2009b). Although real farm data has the advantage of being an actual farm system and its specifics, there are shortcomings to this approach. Firstly, the quality of farm data is often poor. Important data in relation to GHG emissions, such as slurry handling dates and application rates, are not routinely collected on many farms and therefore much data is retrospective and open to error. Secondly, the relevance of the data from any single farm is not always clear for the region or farming system they are designed to represent. Finally, the transitory nature of farming systems, particularly beef farming systems where selling/buying dates are often on an *ad hoc* basis and production cycles run into two or more years, means that a number of years data are generally required before a 'steady state' farm system can be satisfactorily determined.

5.2.2. Boundaries assumed and emission factors applied

With regard to boundaries and GHG emissions, all studies included on-farm and pre-farm (*i.e.*, associated with purchased inputs) emissions. Indirect emissions resulting from volatilisation and deposition of NH_3 and leaching of N_2O-N were also typically included, although this was not specified in the studies of Casey and Holden (2005a, 2006a). Three studies (Williams et al., 2006; Basset-Mens et al., 2009b; Rotz et al., 2010) included machinery while Williams et al. (2006) and Basset-Mens et al. (2009b) also included buildings related emissions. A further two studies included post farm gate emissions associated with animal processing to human edible products (Gerber et al., 2010; Peters et al., 2010). The analyses of Williams et al. (2006), Basset-Mens et al. (2009b) and Nguyen et al. (2010) do not provide a breakdown of emission source burdens making it impossible to quantify the contribution of machinery and buildings to total emissions and provide a consistent comparison with other studies. Gerber et al. (2010) does provide a breakdown of emission sources and specifies that 93% of total GHG emissions from milk production globally occur up to the 'farm gate'.

Inconsistencies in assumed boundaries are paralleled by differing emission factor sources. Emission factors used are primarily taken from IPCC guidelines, although five studies (Haas et al., 2001; Casey and Holden, 2005a, 2006a; Williams et al., 2006; Thomassen et al., 2008b) used published emission factors which were considered more relevant for the particular farm conditions or system modelled. The impact of using different emission factors was highlighted by Olesen et al. (2006) in modelling European dairy systems. In this analysis, three sets of emission factors were applied for CH_4 and N_2O emissions, being default literature sourced emission factors, IPCC Tier 1 values and IPCC Tier 2 values. Results indicated that the standard deviation for the different methods was about 19% of emission values. This was mirrored by the analysis of Gibbons et al. (2006), who showed large differences between calculated enteric CH_4 production when using IPCC estimates or when using regression equations determined experimentally at the University of Reading in the UK. In general, emission factors used in the analysis is a key factor to consider when comparing study findings.

5.2.3. Co-product allocation approaches

Allocation of GHG emissions to co-products can also result in important differences among model outcomes. Since separation and processing of co-products of beef production systems (*e.g.*, meat, hide, tallow) typically occurs post farm gate, and the boundaries of the studies do not extend beyond the farm gate (with the exception of Peters et al., 2010), no allocation is used. In the study of Peters et al. (2010), where post farm gate emissions were considered, all on farm emissions were allocated to beef meat. Where multiple enterprise farms are modelled (Cederberg and Stadig, 2003; Williams et al., 2006; Pelletier et al., 2010; Peters et al., 2010) a biological cause or economic allocation was used to allocate farm level emissions.

The issue of co-product allocation is more important for dairy than for beef systems given that the co-products, beef from cull cows and progeny surplus to replacements, are produced within the boundary of the studies. Weidema and Schmidt (2010) argued that system expansion is most appropriate for these scenarios since mass and energy balances remain intact. However, given that system expansion takes into account the 'avoided burden' of a secondary co-product, in this case beef from an alternative production stream, this approach may be false given that beef consumption globally is increasing (FAO, 2010). Rotz et al. (2010) also contended that system expansion in dairy production systems creates an unfair bias in favour of dairy systems since dairy calf to beef emissions are assumed to originate from the beef cow production systems. Given that the beef produced in dairy systems is largely unavoidable, these authors argue that this method of allocation is not appropriate and that economic allocation is preferable.

The sensitivity of results to alternative allocation approaches was presented by Cederberg and Stadig (2003) and Casey and Holden (2005a). The former investigated implications of four alternative allocation approaches being, no allocation, economic allocation, biological cause allocation and systems expansion. The choice of allocation approach had a substantial effect on the GHG emissions burden, and reduced the emissions burden allocated to milk by 8, 15 and 37% where economic, biological cause and system expansion allocation, respectively, were used relative to no allocation. Casey and Holden (2005a) used no allocation, economic allocation and mass allocation approaches and the emissions burden to milk reduced by 3 and 15% for economic and mass allocation approaches, respectively. It is apparent that the allocation approach has a substantial impact on the relative burden of emissions allocated to milk and beef.

6. Conclusions

The IPCC methodologies provide a set of generalised guidelines for compiling and reporting national inventories and, as such, provide a transparent and consistent framework for comparing national GHG emissions at various times. However, limitations with respect to this national and sectoral approach undermine the usefulness of the methodology for modelling at the farm level. In this respect, whole farm modelling is widely employed for farm level GHG emissions modelling. Despite the increased flexibility and sensitivity of these models to capture farm level activity, concerns and variation relating to data sources, boundaries (*i.e.*, which emission sources are included), emission factors and allocation approaches exist, and they limit direct comparison among studies. Thus, it is critical that assumptions made in this regard are clearly outlined. Furthermore, uncertainty analysis has a key role to play given the range of GHG emission factors reported and the inherent uncertainty involved in agricultural processes. Nevertheless, in terms of developing and assessing mitigation policies to reduce GHG emissions from livestock systems, whole farm approaches provide the most robust and comprehensive approach to developing and implementing effective strategies.

Conflict of interest

None.

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