How do farm models compare when estimating greenhouse gas emissions from dairy cattle production?

N.J. Hutchings, Ş . **Özkan Gülzari**, M. de Haan and D. Sandars

Models used

The order of the models is alphabetical, with no intention to rank them.

DairyWise

The DairyWise model includes all major subsystems of a dairy farm. The central component of DairyWise is the FeedSupply model, which meets the herd requirements for energy and protein, using home-grown feeds (grazed or cut grass, forage crops e.g. maize), maize silage and imported feed. The deficit between requirements and supply is imported as concentrates and roughage (Alem and Van Scheppingen, 1993, Schroder *et al.*, 1998, Zom *et al.*, 2002, Vellinga *et al.*, 2004, Vellinga, 2006, Schils *et al.*, 2007). Methane (CH4), nitrous oxide (N2O), and carbon dioxide (CO2) emissions are calculated in the sub-model greenhouse gas (GHG) emissions, which uses the emission factors from the Dutch emission inventories (Schils *et al.*, 2006). Methane emissions from enteric fermentation are calculated with the Tier 3 model developed by,using different emission factors for concentrate, grass products, and maize (*Zea mays* L.) silage. The emission factors used to calculate CH4 emissions from manure storage are those used in the MITERRA model (Velthof *et al.*, 2007). Direct N2O emissions are related to manure management, nitrogen (N) excreted during grazing, manure application, fertilizer use, crop residues, N mineralization from peat soils, grassland renewal, and biological N fixation. The emission factors are specified according to soil type and ground water level, with generally higher emissions on organic soils and wetter soils. Indirect N2O emissions resulting from the partial denitrification of nitrate (NO3-) resulting from the oxidation of reduced N forms are calculated based on ammonia (NH3) volatilization and NO3- leaching. The emissions of NH3 volatilised are calculated separately for animal housing, manure storage and field-applied manure and fertiliser. Nitrate leaching to ground water was calculated for sandy soils according to the NO3- leaching model of (Vellinga *et al.*, 2001). The amount of NO3- leached was related to the amount of soil mineral nitrogen (SMN) to a depth of 1 meter at the end of the growing season and soil type. The ground water table determined the partitioning of SMN in NO3- leaching and denitrification. The lower the groundwater table, the higher the proportion of NO3- leaching. For grassland, a basic SMN was calculated from the difference between applied and harvested N. In the case of grazing, additional SMN was calculated from urine excretions.

FarmAC

The FarmAC model simulates the flow of carbon (C) and N on arable and livestock farms, enabling the quantification of GHG emissions, N losses to the environment and C sequestration in the soil. It was constructed as part of the EU project AnimalChange (http://www.animalchange.eu/). It is intended to be applicable to a wide range of farming systems across the globe. The model is parameterised separately for each agro-climatic zone.

A static livestock model is used in which the user defines the average annual number of dairy cows, heifers and calves on the farm and the feed ration (including grazed forage). Ruminant livestock production is modelled using a simplified version of the factorial energy accounting system described in CSIRO (2007). Protein supply limitations on production are simulated using an animal N balance approach. Losses of C in CO2 and CH4 are simulated using apparent feed digestibility and IPCC (2006) Tier 2 methods, respectively. Carbon and N in excreta are partitioned to grazed pasture in the same proportion as grazed DM contributes to total DM intake, with the remainder partitioned to the animal housing. Tier 2 methodologies are used for simulating flows in animal housing (CO2 and NH3), manure storage (CO2, CH4, N2O, N2 and NH3) and for N2O, N2 and NH3 emissions from fields. A dynamic model is used to simulate crop production and nutrient flows in the field. The dynamics of soil C are described using the C-Tool model (Taghizadeh-Toosi et al., 2014). A simple soil water model (Olesen and Heidmann, 1990) is used to simulate soil moisture content and drainage. Soil organic N degradation follows C degradation. Mineral N is not chemically speciated. The pool of mineral N is increased by the net mineralisation of organic N and by inputs of fertiliser and manure. It is depleted by leaching, denitrification and crop uptake. The N2O emissions associated with the modelled NH3 volatilisation and NO3- leaching were calculated using IPCC (2006). Crop production is determined by a potential production rate, moderated by N and water availability. The user determines the type, amount and timing of fertiliser and manure applications to each crop.

HolosNor

HolosNor was developed as a farm-scale model to calculate the GHG emissions produced from combined dairy and beef productions systems (Bonesmo *et al.*, 2013) in Norway. It is based on the Canadian Holos model (Little, 2008) utilising the IPCC methodology (IPCC, 2006) modified for Norwegian conditions. The GHGs accounted for in HolosNor are CH4 emissions from enteric fermentation and manure, direct N2O emissions from agricultural soils, indirect N2O emissions resulting from NO3- leached, N in run-off and NH3 volatilised. Both direct and indirect N2O emissions include emissions from manure and synthetic fertiliser applications in soils.

The calculations of all emissions are explained in Bonesmo *et al.* (2013) in details based on Tier 2 approach. Here, only the modification made to the model and input parameters to run the model are described. The ration consisted of grazed grass, grass silage (maize silage in the grass and maize system) grown on farm and concentrates. There was no crop production on the farm. Therefore, concentrates consisting of barley and soybean meal were purchased outside the farm. The GHG emissions associated with production of purchased concentrates were calculated from the mix of barley and soya that could provide the amount of energy and protein in the purchased concentrate (Bonesmo *et al.*, 2013). The amount of concentrates required was calculated using a regression model (Åby *et al*., 2015) based on concentrate intake and forage requirement for different levels of milk production, as described in Volden (2013). Total net energy requirement (NE; MJ cow-1 day-1) was calculated based on the IPCC (2006) recommendations considering maintenance, activity, lactation and pregnancy requirements. Total NE requirement was then converted to DM by taking into account the energy density of the feeds used (6 and 6.5 MJ NE (kg DM)-1 for grass and maize silages, respectively) (http://feedstuffs.norfor.info/). Silage requirement per cow was then calculated by multiplying the total DM requirement by the silage proportion in the ration. By dividing the total farm silage requirement by the potential DM yield given as an input parameter (but corrected for fresh weight and feeding losses), the area to grow silage was computed. The remainder area was allocated for grazing. In the maize scenario, the above and below ground N residue concentration, yield ratio, and above and below ground residue rations were adjusted according to Janzen *et al.* (2003). Methane conversion factor for the warm climate was also adjusted according to IPCC guidelines, as the default values represented the cool climate (IPCC, 2006). In calculating the soil and weather data as one of the required input data, a 45% clayey soil for the Netherlands was found to be outside the normal variation, and therefore the clay content of 35% was applied (A. O. Skjelvåg, Ås, 2016, personal communication).

SFARMMOD

The Silsoe whole-FARM MODel is a linear programme (LP) that maximises long-run farm profit. The concept and structure of the arable farm model are described in Audsley (1981) with the mathematical structure fully described in Annetts and Audsley (2002). The latter paper details the extensions to model mixed arable and livestock systems. The main focus of the environmental burdens concerns the N cycle. Methane emissions were also included, but only from animal agriculture. Sources of information include inventories (Pain *et al.*, 1997, Sneath *et al.*, 1997, Chadwick *et al.*, 1999) and experimental data and mechanistic models (Scholefield *et al.*, 1991, Bouwman, 1996, Smith *et al.*, 1996, Chambers *et al.*, 1999, MAFF, 2000). Some could be used directly (e.g. indirect N2O emissions associated with NH3 volatilisation from animal houses), but others required considerable adaptation to meet the long-term needs of the LP framework (e.g. NO3- leaching) and to ensure that nutrient cycles are closed with no change in N storage in the soil (Williams *et al.*, 2002, Sandars *et al.*, 2003, Williams *et al.*, 2003). Feed is calculated by a linear programme feed ration using dry matter, energy and protein with a 2-week timestep and annual steady state. The model optimises farm cropping which includes grass (grazing and silage) and forage maize silage for the livestock feed. Concentrates with differing energy and protein contents are available to supplement the differing forages.

Product-based emission intensity and allocation methods

The focus of the current work was the extent to which model intercomparison could contribute to quality control of estimates of GHG emissions used in connection with the ESR. The use of emission intensities expressed in kg CO2e (kg ECM)-1 in the main text of the paper was a device to remove the effect of differences between models in the efficiency of production at the animal level, so that other differences could be identified and investigated. However, we recognise that interest by commercial companies (Tesco PLC, 2016) and by those wishing to gain a holistic overview of the environmental impact of product mean that there may be some readers who are interested in the product-base emission intensities *per se*.

The calculation of product-based GHG emission intensities is commonly estimated using Life Cycle Analysis. Such analyses include pre- and post-farm GHG emissions. These were not included in the current study and nor were any on-farm emissions not included in the ESR. Since C sequestration in the soil could not be simulated by all the models in the study, this was excluded from the comparison (models that could simulate C sequestration were run to steady state, so there was no net change in the C sequestered in the soil).

When focussing on the product-based GHG emissions, it is common in systems in which there is more than one product to use one or more allocation methods to partition the environmental impact between products (e.g. Casey and Holden, 2005). In the scenarios presented in the main text, there are two products; milk and meat. The choice of allocation method can have a considerable effect on the resulting emission intensity (Rice *et al.*, 2017). No allocation method was employed when calculating the values reported in the main text of this paper. This is equivalent to following FAO (2010) and considering the production of calves for female replacement as an essential feature of the milk production system. Since male calves are exported at birth, the remaining meat production is minimal. Alternatively, using protein production as an allocation criterion, the GHG emissions would be partitioned 82% to milk and 18% to meat production.

Supplementary references

Alem vG and Van Scheppingen A 1993. The development of a farm budgeting program for dairy farm*.* In Proceedings XXV CIOSTA-CIGR V Congress, pp. 326-331.

Annetts JE and Audsley E 2002. Multiple objective linear programming for environmental farm planning. Journal of the Operational Research Society 53, 933-943.

Audsley E 1981. AN ARABLE FARM MODEL TO EVALUATE THE COMMERCIAL VIABILITY OF NEW MACHINES OR TECHNIQUES. Journal of Agricultural Engineering Research 26, 135-149.

Bonesmo H, Beauchemin KA, Harstad OM and Skjelvåg AO 2013 Greenhouse gas emission intensities of grass silage based dairy and beef production: a systems analysis of Norwegian farms. Livestock Science 152, 239-252

Bouwman AF 1996. Direct emission of nitrous oxide from agricultural soils. Nutrient Cycling in Agroecosystems 46, 53-70.

Chadwick DR, Sneath RW, Phillips VR and Pain BF 1999. A UK inventory of nitrous oxide emissions from farmed livestock. Atmospheric Environment 33, 3345-3354

.

Chambers BJ, Lord EI, Nicholson FA and Smith KA 1999. Predicting nitrogen availability and losses following application of organic manures to arable land: MANNER. Soil Use and Management 15, 137-143.

CSIRO 2007. Nutrient Requirements of Domesticated Ruminants. CSIRO PUBLISHING.

IPCC 2006. 2006 IPCC Guidelines for National Greenhouse Gas Inventories, Prepared by the National Greenhouse Gas Inventories Programme. In Eds. S Eggleston, L Buendia, K Miwa, T Nagara and K Tanabe), Japan.

Janzen HH, Beauchemin KA, Bruinsma Y, Campbell CA, Desjardins RL, Ellert BH and Smith EG 2003. The fate of nitrogen in agroecosystems: An illustration using Canadian estimates. Nutrient Cycling in Agroecosystems 67, 85-102.

Little S 2008. Holos, a Tool to Estimate and Reduce Greenhouse Gases from Farms: Methodology & Algorithms for Version 1.1.x. Agriculture and Agri-Food Canada, pp169, <http://publications.gc.ca/pub?id=9.691658&sl=0>, Accessed 3 November 2017.

MAFF 2000. Fertiliser recommendations for agricultural and horticultural crops (RB209) In MAFF, London, UK.

Pain B, Misselbrook T, Jarvis S, Chambers B, Smith K, Webb J, Phillips V, Sneath R and Demmers T 1997. Inventory of ammonia emission from UK agriculture. In MAFF Contract WA0630, London, UK.

Tesco PLC 2016. Our carbon footprint. <https://www.tescoplc.com/tesco-and-society/sourcing-great-products/reducing-our-impact-on-the-environment/our-carbon-footprint/>. Accessed 14 February 2016.

Rice P, O'Brien D, Shalloo L and Holden NM 2017. Evaluation of allocation methods for calculation of carbon footprint of grass-based dairy production. Journal of Environmental Management 202, 311-319.

Sandars DL, Audsley E, Canete C, Cumby TR, Scotford IM and Williams AG 2003. Environmental benefits of livestock manure management practices and technology by life cycle assessment. Biosystems Engineering 84, 267-281.

Schils RLM, Verhagen A, Aarts HFM, Kuikman PJ and Sebek LBJ 2006. Effect of improved nitrogen management on greenhouse gas emissions from intensive dairy systems in the Netherlands. Global Change Biology 12, 382-391.

Schils RLM, de Haan MHA, Hemmer JGA, van den Pol-van Dasselaar A, De Boer JA, Evers AG, Holshof G, van Middelkoop JC and Zom RLG 2007. DairyWise, a whole-farm dairy model. Journal of Dairy Science 90, 5334-5346.

Scholefield D, Lockyer DR, Whitehead DC and Tyson KC 1991. A model to predict transformations and losses of nitrogen in UK pastures grazed by beef-cattle. Plant and Soil 132, 165-177.

Schroder JJ, Neeteson JJ, Withagen JCM and Noij IGAM 1998. Effects of N application on agronomic and environmental parameters in silage maize production on sandy soils. Field Crops Research 58, 55-67.

Smith JU, Bradbury NJ and Addiscott TM 1996. SUNDIAL: a PC-based system for simulating nitrogen dynamics in arable land. Agronomy Journal 88, 38-43.

Sneath R, Chadwick D, Phillips V and Pain B 1997. A UK inventory of methane emissions from farmed livestock. In Report to MAFF on Project WA0605 and WA0605, London, UK.

Vellinga TV 2006. Management and nitrogen utilisation of grassland on intensive dairy farms. Wageningen.

Vellinga TV, Van der Putten AHJ and Mooij M 2001. Grassland management and nitrate leaching, a model approach. Netherlands Journal of Agricultural Science 49, 229-253.

Vellinga TV, Andre G, Schils RLM and Oenema O 2004. Operational nitrogen fertilizer management in dairy farming systems: identification of criteria and derivation of fertilizer application rates. Grass and Forage Science 59, 364-377.

Velthof GL, Oudendag DA and Oenema O 2007. Development and application of the integrated nitrogen model MITERRA-EUROPE. Alterra-rapport 1663.1, <http://content.alterra.wur.nl/Webdocs/PDFFiles/Alterrarapporten/AlterraRapport1663.1.pdf>. Accessed 3 November 2017.

Volden H 2013. Yield level in milk production – feed and area requirement. In Bioforsk Fokus Eds. E Fløistad and M Günther), pp. 42-44.

Williams AG, Sandars DL and Audsley E 2002. Framework to evaluate farm practices to meet multiple environmental objectives. In Final report for Defra for Project Number WA 0801, London, UK.

Williams AG, Sandars DL, Annetts JE, Audsley E, Goulding KW, Leech P and Day W 2003. A framework to analyse the interaction of whole farm profits and environmental burdens*.* In Proceedings of EFITA 2003 4th Conference of the European Federation for information technology in Agriculture, Food and Environment, Debrecen, Hungary, pp. 492–498.

Zom R, Van Riel J, André G and Van Duinkerken G 2002. Predicted intake with the cow model. In Praktijkonderzoek Veehouderij, Lelystad, The Netherlands.

Åby BA, Crosson P, Aass L and Harstad OM 2015. Effects of increased milk yield per cow on GHG emissions from milk and beef production. Proceedings at European Association of Animal Production, Warsaw, Poland, 31 August–4 September 2015.