

RESEARCH REVIEW

Greenhouse gas emissions from dairy manure management: a review of field-based studies

JUSTINE J. OWEN and WHENDEE L. SILVER

*Department of Environmental Science, Policy & Management, University of California, 130 Mulford Hall #3114, Berkeley, CA 94720, USA***Abstract**

Livestock manure management accounts for almost 10% of greenhouse gas emissions from agriculture globally, and contributes an equal proportion to the US methane emission inventory. Current emissions inventories use emissions factors determined from small-scale laboratory experiments that have not been compared to field-scale measurements. We compiled published data on field-scale measurements of greenhouse gas emissions from working and research dairies and compared these to rates predicted by the IPCC Tier 2 modeling approach. Anaerobic lagoons were the largest source of methane (368 ± 193 kg CH₄ hd⁻¹ yr⁻¹), more than three times that from enteric fermentation (~ 120 kg CH₄ hd⁻¹ yr⁻¹). Corrals and solid manure piles were large sources of nitrous oxide (1.5 ± 0.8 and 1.1 ± 0.7 kg N₂O hd⁻¹ yr⁻¹, respectively). Nitrous oxide emissions from anaerobic lagoons (0.9 ± 0.5 kg N₂O hd⁻¹ yr⁻¹) and barns (10 ± 6 kg N₂O hd⁻¹ yr⁻¹) were unexpectedly large. Modeled methane emissions underestimated field measurement means for most manure management practices. Modeled nitrous oxide emissions underestimated field measurement means for anaerobic lagoons and manure piles, but overestimated emissions from slurry storage. Revised emissions factors nearly doubled slurry CH₄ emissions for Europe and increased N₂O emissions from solid piles and lagoons in the United States by an order of magnitude. Our results suggest that current greenhouse gas emission factors generally underestimate emissions from dairy manure and highlight liquid manure systems as promising target areas for greenhouse gas mitigation.

Keywords: dairy, emissions, emissions modeling, greenhouse gas, IPCC, livestock, manure

Received 2 April 2014 and accepted 1 July 2014

Introduction

Animal agricultural currently accounts for 20% of non-CO₂ greenhouse gas (GHG) emissions globally (EPA, 2012). The majority of these emissions are derived from enteric fermentation by ruminants, especially beef and dairy cattle; however, as livestock agriculture is industrialized, manure management contributes an increasingly large proportion of GHG emissions. This is particularly the case for dairy production which, unlike beef production, occurs predominantly on feedlots in most industrialized countries. In the United States, approximately 43% of CH₄ emissions from dairies were from manure management (USDA, 2011), whereas in California, the state with the greatest dairy production, 54% of dairy CH₄ was estimated to come from manure management (CARB, 2011a). Manure management can also be an important source of nitrous oxide (N₂O) emissions, accounting for an estimated 5% of global (EPA, 2012) and US (EPA, 2013a) N₂O emissions.

Modeling estimates suggested that N₂O emissions from manure management globally played a dominant role in the atmospheric increase in N₂O over the last 140 years (Davidson, 2009).

Emissions from dairy manure management are challenging to measure and model due to the variability in management systems. Greenhouse gas sources associated with manure management include solid and liquid manure storage systems and dairy surfaces in corrals and barns (Fig. 1). To facilitate estimates of GHG emissions from dairies, the Intergovernmental Panel on Climate Change (IPCC) developed a Tier 2 model using emission factors based on manure composition, manure production rates, biogeochemical reaction rates, temperature, pH, and moisture content (IPCC, 2006). Emission factors developed by the IPCC for dairies were largely based on a few lab or pilot-scale studies (IPCC, 2006; Sedorovich *et al.*, 2007; Chadwick *et al.*, 2011). However, the relationship between small-scale studies and actual field emissions is poorly constrained, with only one study making a qualitative comparison (Jungbluth *et al.*, 2001). Comparing emission rates calculated using the Tier 2

Correspondence: Justine J. Owen, tel. +1 510 642 3874, fax +1 510 643 5438, email: justine_owen@berkeley.edu

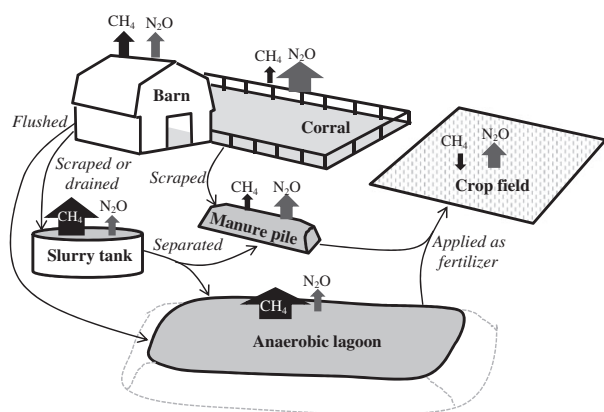


Fig. 1 Sources of N_2O and CH_4 on dairies. Thin arrows indicate movement of manure between locations. Thick arrows indicate relative emission rate. Hardstandings are not shown but have negligible emissions.

model with field measurements provides a valuable test of current emission factors.

Previous reviews of animal agriculture emissions have pooled a variety of livestock systems and scales of studies, i.e. laboratory, pilot, and field scales (Jungbluth *et al.*, 2001; Monteny *et al.*, 2001; Sedorovich *et al.*, 2007; Chadwick *et al.*, 2011; Borhan *et al.*, 2012). However, GHG emissions from dairies likely differ from other livestock industries due to differences in animal and manure management. For example, California dairy feedlots had 1.84 million milk cows and 0.78 million dairy heifers in 2009 which produced 16.4 million kg of volatile solids d^{-1} and 0.92 million kg N d^{-1} , over 55% of which was managed in anaerobic lagoons (CARB, 2011b). In contrast, California beef feedlots had 0.46 million heifers and steers which produced just 0.85 million kg volatile solids d^{-1} and 0.07 million kg N d^{-1} , 1% of which was managed as liquid slurry (CARB, 2011b). Accurately estimating the GHG production from this large stock of dairy manure is critical for designing successful climate change mitigation programs.

The goals of this study were to synthesize a global dataset on GHG emissions from manure management on dairies, compare the data with modeled values, and identify the greatest mitigation opportunities. Published field measurements of CH_4 and N_2O emissions from on dairy manure management on working and research dairies globally were reviewed. Carbon dioxide emission rates were also compiled but because they are not considered to contribute to climate change (IPCC, 2006; with some contention, e.g. Goodland, 2014) they are not discussed further. We compared mean emission rates from the field data with values calculated using the IPCC Tier 2 model to identify discrepancies between measured and modeled values. We

used the field data to derive revised emission factors and used these to calculate new emissions estimates for dairy GHG emissions for the United States and Europe.

Sources of GHG on dairies

Many areas on dairies are potential sources of GHGs, in addition to the direct emissions from cows (Fig. 1). Manure is stored in solid or liquid form. Solid manure piles are composed of the solids scraped from dairy surfaces (manure and bedding) and/or the solids separated from slurry. They are heterogeneous in composition and can have both aerobic and anaerobic zones within the piles, depending on moisture content and management practices. Liquid manure systems were split into two groups: (i) anaerobic lagoons and (ii) slurry tanks and settling ponds, following the approach of the IPCC (2006). Lagoons are earthen and hold the liquid fraction after mechanical or gravity-driven separation of the manure plus wash water. They are not stirred and anaerobic conditions develop rapidly. Slurry tanks and settling ponds are filled with unseparated, minimally diluted manure. Slurry manure has more solids than anaerobic lagoon contents, some of which typically floats on the surface and forms a surface crust. The crust is important because it provides a substrate that spans anaerobic and aerobic environments, where N_2O production and CH_4 oxidation can both occur (Petersen *et al.*, 2005; Petersen & Sommer, 2011). Anaerobic digesters are another liquid manure management system, but no studies have attempted to measure greenhouse gas emissions from functioning anaerobic digesters. Biogas production from anaerobic digesters has been widely studied, but the literature has focused on the potential reduction compared to other manure storage, rather than quantifying greenhouse gas emissions from the anaerobic digester systems themselves. Emissions from digesters are likely to be dependent upon the type of system and operation practices (e.g. retention times, effectiveness of seals in preventing gas escape, and composition of material entering the digester) (Massé *et al.*, 2011; Tauseef *et al.*, 2013).

Corrals included dry lots, loafing pens, and hardstandings. Dry lot corrals are dirt-floored pens in which manure is deposited and occasionally scraped into piles and/or removed. Loafing pens are commonly dirt-floored and spread with some sort of bedding material, often dried manure solids. Milk cows are in loafing pens only when they are not in the milking parlor, dry lots, or freestalls, thus, loafing pens do not accumulate much manure. Some pasture-based dairies use standoff pads to hold cows during wet periods when the cows cannot be on the pastures. These are small corrals in

which a thick (60–100 cm) layer of sawdust and bark chips is laid over plastic sheeting (Luo & Sagggar, 2008). The sheeting allows the leachate from the pad to be collected and treated in liquid storage systems. Hard-standings are areas with solid surfaces, such as concrete, which may be used as corrals or as temporary holding pens, depending on their size and location on the dairy.

Barns were measured as entire barns or only barn floors, depending on measurement approach. Measurements of whole barns include pens and/or freestalls, manure removal and feeding alleys, and often the cows themselves. Barn floors are heterogeneous and typically have paved or slatted-floor areas for livestock movement, farmer access, and manure management, as well as stalls or pens with some sort of soft bedding where the cows can rest. Emissions for entire dairies were reported by two studies and were also included.

Field measurement data compilation

Thirty-eight studies met our criteria (Table 1), most of which were located in North America and Europe (Fig. 2; Table S1). Emission rates were measured using flux chambers or micrometeorological techniques. Measurement techniques varied by dairy source area, which was expected given the different spatial scales (piles vs. whole barns) or materials (liquid vs. solid) involved. Measurements were typically carried out every 1–2 months over 1–5 days for up to a year. Data compiled from the studies included farm characteristics such as the surface area of the pens and lagoons, and number of cows, as available; measurement and gas analysis technique; sampling duration and frequency; and climate data as mean annual temperature (MAT), mean annual precipitation (MAP), and temperature during sampling (Tables S2–S8). A difficulty in comparing literature data was the difference in, or lack of, information reported. When possible we remedied this by contacting the authors or providing reasonable estimates of missing information. Missing MAP and MAT data were estimated using data from the nearest city on www.worldclimate.com. Air temperature during sampling periods ('sampling temperature') was estimated using either the monthly averages from www.worldclimate.com or the almanac feature on www.wunderground.com. Methane fluxes that included enteric fermentation-derived emissions from barns, corrals, or whole dairies were corrected for enteric emissions by subtracting the IPCC regional estimate for enteric fermentation (IPCC, 2006). Specifically, we used $128 \text{ kg hd}^{-1} \text{ d}^{-1}$ for North American studies and $117 \text{ kg hd}^{-1} \text{ d}^{-1}$ for European and modern, high-producing Chinese dairies (rather than the default of

$68 \text{ kg hd}^{-1} \text{ d}^{-1}$ for Asia, which assumes low-producing cows on small farms).

Most studies included measurements of the same area at different times (e.g. seasonally) and/or measurements from areas in which some management component was different (e.g. mixed vs. static manure piles, barns with different flooring and scraping mechanisms). Each permutation was included in the compilation. The mean emission rate for a given dairy area was calculated by first averaging the emission rates compiled from each article, then averaging those values, such that n is the number of studies rather than the number of measurements. This method avoided weighting the mean toward studies, management practices, and measurement techniques with more measurements. Some studies used climate data to extrapolate between measurements to calculate an annual emission rate. We included these annual estimates in the appendices, but they were excluded from the calculation of mean emission rates and the statistical analyses. Statistical analyses were performed using JMP 10.0.2 (SAS Institute, 2012). Correlations between GHG emission rates, climate variables, cow populations, manure volume, and other variables in Tables S2–S8 were explored using multiple linear regressions, with statistical significance determined as $P < 0.10$.

Measurement technique may have affected emissions measured from all manure management systems; large footprint techniques generally measured higher CH_4 and N_2O emissions than studies using dynamic or static chambers, with the exceptions of CH_4 from anaerobic lagoons and corrals. The varied composition and oxygen availability of manure stores creates CH_4 and N_2O emission hotspots in space and time, which can be missed by smaller footprint techniques (Parkin & Kaspar, 2004; Sommer *et al.*, 2004). Concurrent measurements using different techniques have not been made (with one exception: Sommer *et al.*, 2004), but are needed to resolve their impact on reported emissions. Because of the uncertainty in the extent of the impact of measurement technique, in the analysis below we calculated mean emissions from each area using all available data. In most cases, this likely produced a conservative estimate.

Emission rates are presented as the mass of trace gas emitted per head per time ($\text{kg trace gas hd}^{-1} \text{ yr}^{-1}$) and per unit area per time ($\text{kg trace gas m}^{-2} \text{ yr}^{-1}$). The discussion focuses on per head numbers for several reasons. The goal of this study was to evaluate our ability to estimate dairy emissions from manure management at regional to global scales; therefore, emissions factors needed to use units that were widely known. Most countries have fairly good estimates of the number of animals present, but estimates of the area of the various

Table 1 Studies included in this review and the sources of greenhouse gases measured by each

| Study | Lagoon | Slurry tank | Manure pile | Compost area | Corrals | Concrete pens | Barn | Whole dairy |
|-------------------------------------|--------|-------------|-------------|--------------|---------|---------------|------|-------------|
| Borhan <i>et al.</i> (2011a) | × | | | × | × | | × | |
| Borhan <i>et al.</i> (2011b) | × | | | | × | | × | |
| Bjorneberg <i>et al.</i> (2009) | × | | | | × | | | |
| Leytem <i>et al.</i> (2011) | × | | | × | × | | | |
| Leytem <i>et al.</i> (2013) | × | | | | | | × | |
| Craggs <i>et al.</i> (2008) | × | | | | | | | |
| Safley and Westerman (1988) | × | | | | | | | |
| Safley and Westerman (1992) | × | | | | | | | |
| Todd <i>et al.</i> (2011) | × | | | | | | | |
| Husted (1994) | | × | × | | | | | |
| Sneath <i>et al.</i> (2006)* | | × | × | | | | | |
| Hensen <i>et al.</i> (2006) | | × | | | | | | × |
| Kaharabata <i>et al.</i> (1998) | | × | | | | | | |
| Khan <i>et al.</i> (1997) | | × | | | | | | |
| VanderZaag <i>et al.</i> (2011) | | × | | | | | | |
| Ahn <i>et al.</i> (2011) | | | × | | | | | |
| Amon <i>et al.</i> (2006) | | | × | | | | | |
| Brown <i>et al.</i> (2002) | | | × | | | | | |
| Gupta <i>et al.</i> (2007) | | | × | | | | | |
| Osada <i>et al.</i> (2001) | | | × | | | | | |
| Sommer <i>et al.</i> (2004) | | | × | | | | | |
| Kaharabata <i>et al.</i> (2000) | | | | | × | | | |
| Luo and Saggar (2008) | | | | | × | | | |
| Ellis <i>et al.</i> (2001) | | | | | | × | | |
| Gao <i>et al.</i> (2011) | | | | | | × | | |
| Misselbrook <i>et al.</i> (2001) | | | | | | × | | |
| Adviento-Borbe <i>et al.</i> (2010) | | | | | | | × | |
| Kinsman <i>et al.</i> (1995) | | | | | | | × | |
| Marik and Levin (1996) | | | | | | | × | |
| Ngwabie <i>et al.</i> (2009) | | | | | | | × | |
| Ngwabie <i>et al.</i> (2011) | | | | | | | × | |
| Samer <i>et al.</i> (2012) | | | | | | | × | |
| Snell <i>et al.</i> (2003) | | | | | | | × | |
| van Vliet <i>et al.</i> (2004) | | | | | | | × | |
| Wu <i>et al.</i> (2012) | | | | | | | × | |
| Zhang <i>et al.</i> (2005) | | | | | | | × | |
| Zhu <i>et al.</i> (2012) | | | | | | | × | |
| McGinn and Beauchemin (2012) | | | | | | | | × |

*Excludes slurry tank work which was pilot-scale.

manure handling systems have not been attempted with few exceptions (Chung *et al.*, 2013). Others (Place & Mitloehner, 2010; O'Brien *et al.*, 2012) have argued that reporting emissions in terms of the mass of milk produced gives a better sense of the GHG-efficiency of production. This is a useful approach for comparing different production systems. However, milk production varies significantly by breed, feed, cow age, and stage in lactation cycle (e.g. McCandlish, 1920; Zimmerman *et al.*, 1991); furthermore, it is not relevant to the emissions from different manure management

approaches in which we are interested here. Few studies (only 11 of the 38 studies included here, with six of the studies measuring whole barns) reported milk production.

To compare the global warming potential (GWP) of the measured areas, N₂O and CH₄ emission rates were converted to 100-year CO₂e emission rates by multiplying by 298 and 34, respectively (Myhre *et al.*, 2013), and summing the two. When other inventories used the older GWPs for N₂O and CH₄ of 310 and 21 (IPCC, 1996), respectively, those emission rates were

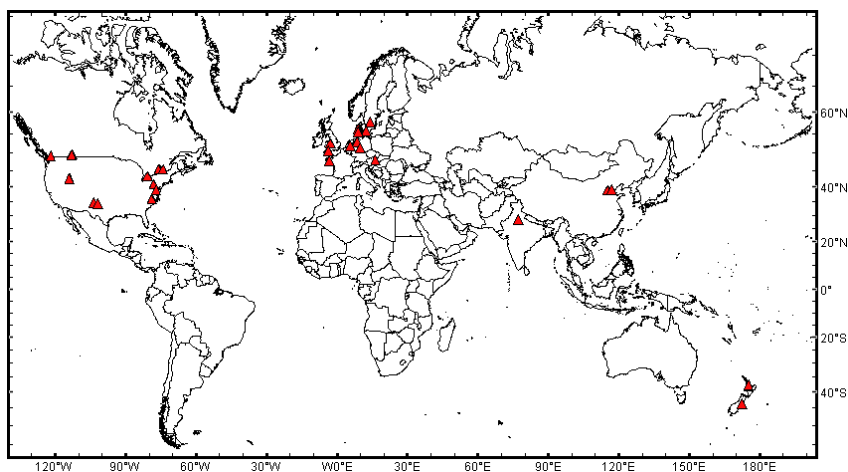


Fig. 2 Geographic distribution of sampling sites, marked as triangles (created in GeoMapApp v. 3.3.8, <http://www.geomapapp.org/>).

recalculated using the revised values to be comparable to ours.

Summary of field measurements

Anaerobic lagoons and slurry systems had the highest per head GWP on dairies, averaging 12.8 ± 7 Mg CO₂e hd⁻¹ yr⁻¹ and 3.5 ± 1.7 Mg CO₂e hd⁻¹ yr⁻¹, respectively (Table 2a). Mean lagoon GWP was about 20 times higher than mean solid manure storage GWP. When expressed on an area basis, lagoons and slurry systems were similar, averaging 703 ± 195 kg CO₂e m⁻² yr⁻¹ and 827 ± 320 kg CO₂e m⁻² yr⁻¹, respectively (Table 2b). These rates were high; for comparison, the highest landfill CH₄ emissions rates reported in Bogner *et al.* (1995) were 248 kg CO₂e m⁻² yr⁻¹, less than half those from liquid manure systems. Barn floors had the lowest GWP (38 ± 7 kg CO₂e hd⁻¹ yr⁻¹) of all the dairy environments studied. Methane emissions were the largest component of total GWP for all sources except for barns and corrals.

Liquid manure storage systems were the greatest source of CH₄, with anaerobic lagoons and slurry stores emitting 368 ± 193 kg CH₄ hd⁻¹ yr⁻¹ and 101 ± 47 kg CH₄ hd⁻¹ yr⁻¹, respectively (Table 2a). Barns were the next largest source with 33 ± 19 kg CH₄ hd⁻¹ yr⁻¹. This was unexpected given that only one study reported subfloor (deep pit) storage and that most others reported relatively frequent scraping and/or flushing that removed substrate for GHG production.

Barns had the greatest N₂O emissions by nearly an order of magnitude, with 10.3 ± 6.2 kg N₂O hd⁻¹ yr⁻¹ (Table 2a), although field data were highly variable (Table S6). Corrals and solid manure piles were the next largest N₂O source with 1.5 ± 0.8 kg N₂O hd⁻¹ yr⁻¹ and 1.1 ± 0.7 kg N₂O hd⁻¹ yr⁻¹,

respectively (Table 2a). Nitrous oxide emissions from anaerobic lagoons and slurry stores were also substantial, with 0.9 ± 0.5 kg N₂O hd⁻¹ yr⁻¹ and 0.3 ± 0.3 kg N₂O hd⁻¹ yr⁻¹, respectively. The relatively large net N₂O flux from liquid manure storage was surprising given the predominantly anaerobic conditions typical of un aerated systems. Nitrogen in liquid manure is mostly in the form of ammonium (NH₄⁺) and organic N (Harter *et al.*, 2002), and though anaerobic lagoons are generally anaerobic, aerobic conditions which could promote denitrification exist at inlets. Other N₂O formation reactions are also feasible, such as denitrification of nitrate (NO₃⁻) produced through annamox [anaerobic NH₄⁺ oxidation, (Mulder *et al.*, 1995; Maeda *et al.*, 2010)], Feammox [anaerobic NH₄⁺ oxidation coupled to Fe reduction, (Yang *et al.*, 2012)], or MnammoX [anaerobic NH₄⁺ oxidation coupled to Mn reduction, (Engström *et al.*, 2005)]. Hardstandings and barn floors, surfaces which were scraped or flushed frequently, had CH₄ and N₂O emissions generally one to three orders of magnitude lower than the other sources. These trends were consistent between the per head and per area data (Table 2b) and showed that the type of storage or surface measured was the greatest factor controlling emission rates.

Methane emissions from soils are known to be temperature dependent (Conrad, 2007) and models often assume that manure CH₄ emissions are positively correlated with MAT (Mangino *et al.*, 2002; IPCC, 2006). Individual field studies observed greater CH₄ emissions in summer and/or with warmer sampling temperatures for manure piles, barns, and whole dairies; however, there was no significant correlation between CH₄ emissions and temperature when all the studies for a given source area were considered. The lack of correlation for liquid systems may be due to the

Table 2 Summary of the means and ranges of N₂O and CH₄ emission rates measured by the studies listed in Table 1, in (a) kg hd⁻¹ yr⁻¹ and (b) kg m⁻² yr⁻¹

| | Emission rate (kg hd ⁻¹ yr ⁻¹) | | | Emission global warming potential† (kg CO ₂ e hd ⁻¹ yr ⁻¹) | | |
|-------------------|---|-------------------------------------|-------------------------------|--|-----------------------|---|
| | Mean* ± SE (n) | | | Mean* ± SE (n) | | |
| | Range | | | From CH ₄ | | |
| | CH ₄ | N ₂ O | CO ₂ | From CH ₄ | From N ₂ O | Total (CH ₄ +N ₂ O) |
| (a) | | | | | | |
| Anaerobic lagoons | 368 ± 193 (9) 4–2814 | 0.9 ± 0.5 (4) 0.004–3.9 | 687 ± 266 (6) 4.8–2400 | 12 510 ± 7334 | 264 ± 131 | 12 775 ± 6699 |
| Slurry stores | 101 ± 47 (6) 0–328 | 0.3 ± 0.3 (3) 0–4.5 | nm | 3422 ± 1601 | 81 ± 76 | 3504 ± 1680 |
| Solid‡ | 13 ± 11 (4) 0–99 | 1.1 ± 0.7 (4) 0.02–7 | 754 ± 695 (2) 59–3546 | 431 ± 372 | 315 ± 196 | 632 ± 470 |
| Corrals§ | –17 ± 24 (6) –128 to 210 | 1.5 ± 0.8 (4) 0.0–12 | 4242 ± 3040 (3) 134–20 292 | –577 ± 844 | 454 ± 272 | –124 ± 1073 |
| Hardstandings§ | 1.2 ± 0.8 (3) –3.8 to 7.1 | 0.0004 ± 0.0001 (2) 0.0001–0.001 | nm | 40 ± 26 | 0.13 ± 0.02 | 40 ± 27 |
| Barn floor | 0.9 ± 0.7 (4) 0–4.4 | 0.03 ± 0.01 (4) 0.001–0.1 | 94 ± 39 (4) 25–250 | 30 ± 22 | 7.5 ± 4.4 | 38 ± 27 |
| Whole barn§ | 33 ± 19 (10) –61 to 289 | 10 ± 6 (3) 0–22 | 7204 ± 5507 (3) 273–35058 | 1120 ± 931 | 3076 ± 3154 | 4197 ± 2496 |
| Whole dairy§ | 96 ± 35 (18) –91 to 350 | nm | nm | 3252 ± 1191 | nm | 3252 ± 1194 |
| (b) | | | | | | |
| | Emission rate (kg m ⁻² yr ⁻¹) | | | Emission global warming potential† (kg CO ₂ e m ⁻² yr ⁻¹) | | |
| | Mean* ± SE (n) Range | | | Mean* ± SE (n) | | |
| | CH ₄ | N ₂ O | CO ₂ | Total (CH ₄ +N ₂ O) | | |
| Anaerobic lagoons | 20 ± 5 (9) 0.3–84 | 0.09 ± 0.05 (4) 0.001–0.4 | 64 ± 34 (6) 2–312 | 703 ± 195 | | |
| Slurry stores | 24 ± 9 (7) 0–74 | 0.05 ± 0.04 (3) 0–0.7 | nm | 827 ± 320 | | |
| Solid‡ | 2.8 ± 0.9 (6) 0–13 | 0.3 ± 0.1 (7) 0.005–1.0 | 268 ± 103 (4) 13–461 | 147 ± 49 | | |
| Corrals§ | –0.8 ± 0.9 (6) –5.2 to 3.8 | 0.03 ± 0.01 (4) 0.001–0.22 | 83 ± 52 (3) 12–365 | –16 ± 35 | | |
| Hardstandings§ | 0.5 ± 0.6 (3) –0.11 to 2.2 | 0.0003 ± 0.0001 (2) 0.000–0.001 | nm | 18 ± 21 | | |
| Barn floor | 0.2 ± 0.2 (4) 0–1.0 | 0.01 ± 0.01 (4) 0–0.4 | 21 ± 10 (4) 5–58 | 9.4 ± 7.0 | | |
| Whole barn§ | 3.4 ± 2.0 (10) –2.8 to 31 | 0.9 ± 0.7 (3) 0–2.3 | 774 ± 578 (3) 39–3713 | 381 ± 277 | | |
| Whole dairy§ | nm | nm | nm | nm | | |

*Mean emissions were calculated by first averaging measurements within studies then averaging across studies (n = number of studies). The exception was the whole dairy measurements which were treated individually and not grouped by study (i.e. n = number of dairies measured).

†Where 1 g CH₄ = 34 g CO₂e and 1 g N₂O = 298 g CO₂e. CO₂ is not included.

‡Excludes data from Gupta *et al.* (2007).

§Where necessary, methane emission rates have been corrected for enteric emissions as described in the text. Carbon dioxide emissions were not corrected for respiration.

limited range of MAT represented by the field studies; all studies but one (Todd *et al.*, 2008) sampled liquid systems that were in regions where MAT was 6–15 °C. Air temperature during sampling had a larger range (–10.6 to 34.4 °C, Tables S2–S3), but overall liquid systems in warm climates were under-represented. Differences in volatile solid content, the other key factor determining CH₄ production (Mangino *et al.*, 2002), may have also confounded any temperature effect. Insufficient data were available to test the effect of volatile solid content on CH₄ emissions within or across studies.

Methane and N₂O emissions were strongly correlated with each other for solid manure piles ($r^2 = 0.73$, $P < 0.001$) and weakly correlated for corrals ($r^2 = 0.26$, $P < 0.08$). This suggests that in solid manure management systems, at least a portion of the N₂O fluxes were derived from denitrification, which requires the same general environmental conditions as methanogenesis (warm temperatures, abundant labile C, anaerobic conditions). Corral CH₄ emissions were negative, indicating soil uptake, in five of 18 cases (Table S4); negative fluxes occurred in late summer when the soils were dry, or in winter when the soils were cold or frozen. The highest corral N₂O emissions were measured in late spring when a combination of warmer temperatures and moist soils likely promoted nitrification and denitrification (Table S4). The lowest values were <50% of the highest emissions and occurred in late fall and winter (Table S4). Despite these seasonal patterns, neither CH₄ nor N₂O emissions from corrals were correlated with temperature and/or precipitation. Leytem *et al.* (2011) measured higher N₂O emission rates from manure piles in warmer months (May and June) than colder ones (September and March, Table S5), but no correlations were found when all manure pile data were pooled.

Specific management practices could have made it difficult to detect a temperature effect if one existed. Mixing solid manure piles resulted in increased CH₄ and N₂O emissions (Yamulki, 2006; Maeda *et al.*, 2010; Ahn *et al.*, 2011; Leytem *et al.*, 2011), contrary to expectations that mixing would aerate the pile and decrease CH₄ production. The addition or accumulation of fresh manure was another source of emissions. Addition of fresh material increased pile emissions (Leytem *et al.*, 2011) and the accumulation of fresh material in corrals was likely one of the most important factors driving positive CH₄ fluxes. Borhan *et al.* (2011) measured greater CH₄ and N₂O emissions from a dry lot corral than from loafing pens (Table S4), probably due to the corrals having a greater influx of fresh manure and localized, high-moisture urine patches. Methane emissions from the brick

hardstanding were relatively high in the summer (Table S7) (Gao *et al.*, 2011), likely because scraping was less frequent compared to other dairies (every 1–4 weeks vs. daily for most other hardstandings, Table S7). Accordingly, Adviento-Borbe *et al.* (2010) observed a significant, positive correlation between CH₄ emissions and manure depth on the barn floor. However, Gao *et al.* (2011) was the only study of hardstandings to use an open path laser rather than flux chambers.

Emission rate modeling

The field measurements provide a test of emission rate models. The IPCC Tier 2 approach models CH₄ emissions (EF_{CH₄}, g CH₄ hd⁻¹ yr⁻¹) based on the volatile solid production by the cows (VS, kg VS hd⁻¹ yr⁻¹), a CH₄ conversion factor (MCF, %) for the manure management practice, and the maximum possible CH₄ production rate from the volatile solids in the manure (B_o , m³ CH₄ kg VS⁻¹):

$$EF_{CH_4} = VS \times MCF/100 \times B_o \times 662 \text{ g CH}_4 \text{ m}^{-3}\text{CH}_4 \quad (1)$$

Volatile solid production by cows can be determined from manure analysis (where volatile solids are the combustible components of solid manure) or estimated based on feed intake rate, digestibility, and dry matter content. No studies reviewed here included all the information necessary to calculate dairy specific VS so we used the IPCC regional values for international data (IPCC, 2006) and averages of state values for the United States (EPA, 2013b).

The IPCC Tier 2 approach models direct N₂O emissions based on annual N excretion rates, which themselves are a function of energy intake by the cows, crude protein content of feed, milk production rate, milk protein content, cow growth, typical animal mass, and an emission factor (EF_{N₂O}, kg N₂O-N kg N excreted⁻¹) (equations 10.31, 10.32, and 10.33 in IPCC, 2006). EF_{N₂O} can be converted into N₂O emission rates equivalent to those measured here (N₂O_D, g N₂O hd⁻¹ d⁻¹, where the subscript D refers to direct emissions) using the typical animal mass (TAM, kg) and country- or region-specific N excretion rates (N_{ex} , kg N 1000 kg TAM⁻¹ d⁻¹):

$$N_2O_D = EF_{N_2O} \times 44/28 \times TAM \times N_{ex} \quad (2)$$

In our calculations, we used a TAM of 600 kg, the default for Western Europe (but similar to the North American default value of 604 kg) (IPCC, 2006). Indirect N₂O emissions, derived from the oxidation of gaseous emissions such as ammonia (NH₃) and nitrous oxides (NO_x), are important for calculating the amount

of N remaining in manure for its use as an organic fertilizer (IPCC, 2006). We did not include indirect N₂O emissions in our calculations, and thus what is reported here should be considered minimum estimates.

We used the field measurement means and Eqns (1) and (2) to derive revised MCFs and EF_{N₂O}s for the source areas. This is the first time broadly applicable, field measurement-derived MCFs and EF_{N₂O}s have been calculated. Some revised MCFs and EF_{N₂O}s were very different from current values.

Comparisons with modeled emissions

Measured vs. modeled CH₄ emissions

The means of the field-measured CH₄ emissions from slurry tanks and barns (deep pit storage) were three times larger than modeled emissions, while the measured CH₄ emissions from solid manure piles and corrals were lower than modeled values, although there was considerable variability in measured values (Table 3; Fig. 3). The modeled CH₄ emissions from the remaining sources (anaerobic lagoons and hardstandings) were within the standard error of the field means or were negligible. Modeled whole dairy CH₄ emissions (calculated using parameters for Western Europe) were slightly lower than the field measurement mean. The default MCFs were within the standard error of the field measurement-derived means except for slurry tanks and whole barns which had larger revised MCFs (Table 3).

The impact of the revised barn/deep pit and slurry store MCFs was evaluated using data on slurry storage in Europe because six of 13 barn studies were conducted in Europe, while slurry studies were distributed in temperate regions globally. We used 1990 and 2011 emissions inventory data for 12 European countries compiled by the United Nations Framework Convention on Climate Change (UNFCCC, 2014). Three of the 15 countries in the dataset were excluded due to lack of data or falling outside the cool MAT temperature zone. The European data did not distinguish between slurry stored in deep pits and tanks or ponds (i.e. one MCF was used for all slurry, that in deep pits and in ponds), whereas we calculated revised MCFs for each system. Thus, revised European slurry CH₄ emissions were calculated using each revised MCF to provide a range. However, deep pit storage is often a temporary holding for slurry that is eventually transferred to slurry tanks or ponds, so the MCF for slurry stores is likely more applicable.

Calculations using the revised deep pit MCF gave total CH₄ emissions from European slurry storage that were less than those using the country-specific slurry

Table 3 Methane emissions modeling inputs and results. VS = volatile solids, MCF = methane conversion factor, B₀ = the maximum possible CH₄ production rate from the volatile solids in the manure

| | CH ₄ emission rate | | | | | | | | |
|--|---|-------------------------|---------|-------------------------|--|----------------------------|---|---|-----------------------|
| | VS (kg VS hd ⁻¹ yr ⁻¹) | VS data source | MCF (%) | MCF data source | B ₀ (m ³ CH ₄ kg VS ⁻¹) | B ₀ data source | Modeled (kg hd ⁻¹ yr ⁻¹) | Field (kg hd ⁻¹ yr ⁻¹) | Field-derived MCF (%) |
| Anaerobic lagoon | 2770 | Average ID, NM, TX, NC* | 74 | Average ID, NM, TX, NC* | 0.24 | Default*† | 326 | 368 ± 193 (9) | 84 ± 44 |
| Slurry stores | 1861.5 | Western Europe† | 11 | MAT=1, with crust† | 0.24 | Western Europe† | 33 | 101 ± 47 (6) | 34 ± 16 |
| Manure pile | 2750 | Average TX and ID* | 5 | Static pile* | 0.24 | Default* | 22 | 13 ± 11 (4) | 2.9 ± 2.5 |
| Corrals | 2800 | Average TX and ID* | 1 | Cool MAT*† | 0.24 | Default*† | 4.4 | -17 ± 24 (6) | -3.8 ± 5.5 |
| Paved surfaces and barn floors | 1861.5 | Western Europe† | 1 | Cool MAT† | 0.24 | Western Europe† | 3.0 | 0.9 ± 0.5 (6) | 0.31 ± 0.16 |
| Brick hardstanding | 1022 | Asia† | 1 | Cool MAT† | 0.13 | Asia† | 0.9 | 1.6 ± 7.7 (1) | 1.8 ± 8.8 |
| Whole barn | 1861.5 | Western Europe† | 3 | Deep pit, cool MAT† | 0.24 | Western Europe† | 8.9 | 33 ± 19 (10) | 11 ± 6.4 |
| Whole dairy (=corral + manure pile + slurry tank + whole barn) | | | | | | | 56 | 96 ± 35 (18) | |

*from (EPA, 2013b).

†from (IPCC, 2006).

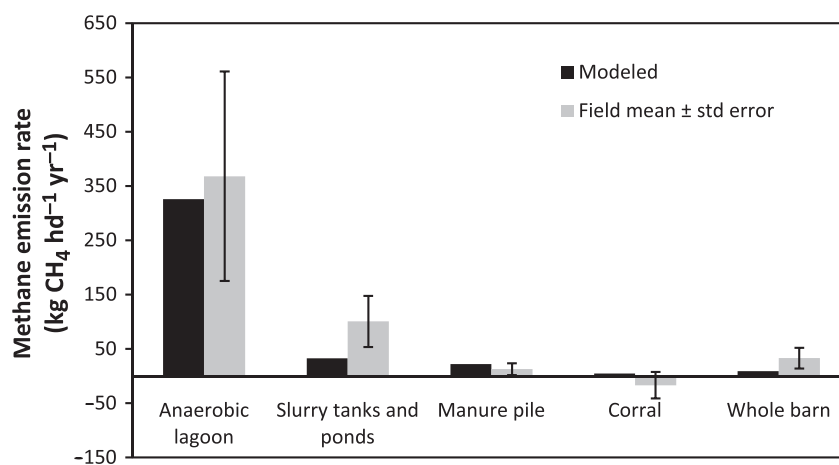


Fig. 3 Comparison of modeled CH₄ emissions and field measurement means and standard errors for the largest CH₄ sources.

MCFs (8.4 ± 4.6 Tg CO₂e yr⁻¹ vs. 15.2 Tg CO₂e yr⁻¹, respectively, Table 4, Fig. 4). However, the revised slurry pond MCF increased CH₄ emissions from slurry for most countries, with total emissions of 25.9 ± 12.2 Tg CO₂e yr⁻¹, a gain of 10.7 Tg CO₂e yr⁻¹ (Table 4, Fig. 4). Increases were greatest for the countries with

the most manure in liquid systems (Denmark, Germany, the Netherlands, Switzerland, and Sweden). We found a similar trend using detailed data for the Netherlands (RIVM *et al.*, 2013), with modeled slurry CH₄ emissions two times larger than those estimated in the current inventory (data and calculations not shown).

Table 4 Comparison of modeled slurry emissions in 13 cool MAT European countries using the 2011 liquid slurry MCFs (and other inputs) for each country (UNFCCC, 2014), the revised deep pit and slurry MCFs from Table 3. The top 5 countries for liquid manure management are Denmark, Germany, the Netherlands, Switzerland and Sweden

| | Dairy cows (1000 hd) | VS (kg DM hd ⁻¹ d ⁻¹) | Bo (m ³ CH ₄ kg ⁻¹ VS) | Liquid fraction (%) | Liquid MCF (%) | Revised MCF (deep pit) (%) | Revised MCF (slurry) (%) | Liquid manure emissions (Gg CH ₄ yr ⁻¹) | | |
|-----------------|----------------------|--|---|---------------------|----------------|----------------------------|----------------------------|--|----------------------------|--------------------------|
| | | | | | | | | Using current MCF | Using revised deep pit MCF | Using revised slurry MCF |
| Austria | 527.39 | 4.27 | 0.24 | 31.61 | 8.7 | 11 ± 6 | 34 ± 16 | 3.6 | 4.5 ± 2.5 | 14.0 ± 6.6 |
| Belgium | 459.78 | 4.10 | 0.24 | 11.54 | 19 | 11 ± 6 | 34 ± 16 | 2.4 | 1.4 ± 0.8 | 4.3 ± 2.0 |
| Denmark | 565.11 | 6.09 | 0.24 | 88.41 | 10 | 11 ± 6 | 34 ± 16 | 17.7 | 19.4 ± 10.6 | 60.0 ± 28.2 |
| Finland | 285.53 | 4.94 | 0.24 | 46.41 | 10 | 11 ± 6 | 34 ± 16 | 3.8 | 4.2 ± 2.3 | 12.9 ± 6.1 |
| France | 3660.68 | 4.12 | 0.24 | 40.87 | 39 | 11 ± 6 | 34 ± 16 | 139.3 | 39.3 ± 21.4 | 121.4 ± 57.1 |
| Germany | 4190.10 | 4.01 | 0.23 | 73.52 | 14.4 | 11 ± 6 | 34 ± 16 | 98.6 | 75.6 ± 41.2 | 233.6 ± 109.9 |
| Ireland | 1086.11 | 2.98 | 0.24 | 28.60 | 39 | 11 ± 6 | 34 ± 16 | 20.9 | 5.9 ± 3.2 | 18.3 ± 8.6 |
| Italy | 1754.98 | 6.37 | 0.14 | 35.03 | 13.9 | 11 ± 6 | 34 ± 16 | 19.0 | 15.0 ± 8.2 | 46.4 ± 21.9 |
| Luxembourg | 40.45 | 4.56 | 0.24 | 34.20 | 39 | 11 ± 6 | 34 ± 16 | 1.4 | 0.4 ± 0.2 | 1.2 ± 0.6 |
| Sweden | 346.50 | 5.33 | 0.24 | 62.23 | 3.5 | 11 ± 6 | 34 ± 16 | 2.3 | 16.0 ± 8.7 | 22.7 ± 10.7 |
| Switzerland | 589.24 | 6.24 | 0.24 | 68.22 | 10 | 11 ± 6 | 34 ± 16 | 14.5 | 40.2 ± 21.9 | 49.4 ± 23.3 |
| The Netherlands | 1469.72 | 4.56 | 0.25 | 90.38 | 17 | 11 ± 6 | 34 ± 16 | 62.2 | 17.1 ± 9.3 | 124.4 ± 58.5 |
| United Kingdom | 1814.00 | 3.61 | 0.24 | 41.00 | 39 | 11 ± 6 | 34 ± 16 | 60.8 | 4.5 ± 2.5 | 53.0 ± 24.9 |
| | | | | | | | All | 446.5 | 246.4 ± 134.4 | 761.7 ± 358.4 |
| | | | | | | | Top 5 liquid | 195.3 | 158.6 ± 86.5 | 490.1 ± 230.6 |
| | | | | | | | All (Tg CO ₂ e) | 15.2 | 8.4 ± 4.6 | 25.9 ± 12.2 |
| | | | | | | | Top 5 liquid | 6.6 | 5.4 ± 2.9 | 16.7 ± 7.8 |
| | | | | | | | (Tg CO ₂ e) | | | |

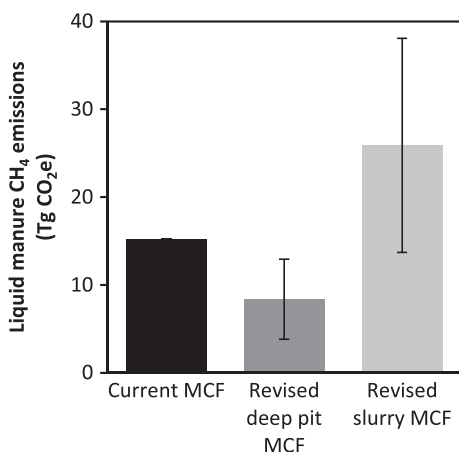


Fig. 4 Comparison of modeled CH₄ emissions for slurry using the current slurry MCF used by the European Union and the revised MCFs for deep pit storage and slurry storage calculated in this study (Table 4).

The uncertainty in slurry MCF has consequences for the evaluation of the European dairy industry's progress in mitigating its GHG emissions. Between 1990 and 2011, the 13 countries considered here decreased the total number of cows by nearly 8 million hd leading to a corresponding decrease in emissions from enteric fermentation by 515 Gg CH₄ (Table 5). The reduction in cows also decreased VS production by 7.8 Tg so there was less manure to manage and produce GHG. However, an increase in the proportion of manure in liquid management in most countries offset some of this decrease in CH₄ production; the current estimates suggest a total net decrease (combined change in enteric and manure management emissions) of 480 Gg CH₄ (Table 5). Using the revised slurry MCF for the 2011 estimates gives a smaller total net decrease of 166 Gg CH₄, with some countries (Denmark, Switzerland, and the Netherlands) having net increases of 30–50 Gg CH₄ rather than decreases (Table 5).

Measured vs. modeled N₂O emissions

Modeled N₂O emissions were less than half of the field measurement means for anaerobic lagoons, solid manure piles, and barns (Table 6). In contrast, the modeled value for slurry stores was greater than the field measurement mean. The other sources had modeled emissions that were within the standard error of the field means or were negligible. The revised EF_{N₂O} values for anaerobic lagoons, manure piles, and barns were larger than the default values, and the slurry EF_{N₂O} was the same as the default (Table 6).

The impact of revised EF_{N₂O} values was evaluated using state-specific data from 2011 for the United

States (Table 7) because eight of the nine anaerobic lagoon studies and five of the ten manure pile studies occurred in the United States or North America; therefore, the revised EF_{N₂O} values should be applicable to this region. The EPA assumed zero N₂O emissions from anaerobic lagoons, whereas the revised EF_{N₂O} gave 1.79 ± 0.90 Tg CO₂e yr⁻¹ (Fig. 5). Nitrous oxide emissions from solid manure piles also increased from 0.51 Tg CO₂e yr⁻¹ to 3.36 ± 2.04 Tg CO₂e yr⁻¹ using the revised EF_{N₂O} (Fig. 5). Combined, the revised values increased manure management N₂O emissions in the United States by more than 4.5 Tg CO₂e yr⁻¹, 25% of the 2011 estimate of 17.3 Tg CO₂e (EPA, 2013a).

Whole barn N₂O emissions varied widely between studies, and the measurements of Leytem *et al.* (2013) and Samer *et al.* (2012) suggested an order of magnitude increase in EF_{N₂O}. They also indicated that barns may be significant, largely unaccounted sources of N₂O from dairies (2–3 times more kg N₂O hd⁻¹ yr⁻¹ than corrals or solid piles). No standard model has been established for calculating N₂O emissions from barns that do not have deep pit manure storage. If we assume that two-thirds of the cows in the 13 European countries in Table 4 were kept in barns (11 million hd) for half of the year, and emitted 1 kg N₂O hd⁻¹ yr⁻¹ (the approximate mean of the measurements by Zhang *et al.* (2005), which was the lowest of the three studies that measured N₂O), then barns emitted 1.64 Tg CO₂e, on the same scale as the revised N₂O emissions from anaerobic lagoons in the United States (Table 7). Using the field measurement mean of 10 kg N₂O hd⁻¹ yr⁻¹ for the calculation increased barn emissions by an order of magnitude (to 16.4 Tg CO₂e), equivalent to the warming potential of slurry system-derived CH₄ in Europe (Table 4). More data are needed to assess if barns are actually such large sources of N₂O.

Discussion and conclusions

Our results highlight potential issues with the application of IPCC Tier 2 models to estimate GHG emissions from livestock manure. Emission factors were typically based on few studies, many of which were not designed for GHG inventory estimation or were small-scale pilot or laboratory experiments, and spanned various livestock systems (Jungbluth *et al.*, 2001; Chung *et al.*, 2013). These approaches are unlikely to accurately approximate field-scale fluxes from manure management in a specific livestock system. Our review of field-based research on dairies suggests that current Tier 2 model parameters are generally underestimating dairy emissions.

Table 5 Comparisons between 1990 and 2011 data for the 13 countries in Table 4 (UNFCCC, 2014). Negative values indicate a decrease from 1990 to 2011. The revised slurry MCF was used to recalculate 2011 emissions, not the revised deep pit MCF, and the 1990 value were not adjusted

| | Cows (1000 hd) | VS production (Gg) | Liquid fraction (%) | Enteric fermentation emissions (Gg CH ₄) | Manure management emissions (default MCF) (Gg CH ₄) | Net emissions (default MCF) (Gg CH ₄) | Manure management emissions (revised MCF) (Gg CH ₄) | Net emissions (revised MCF) (Gg CH ₄) |
|-----------------|-------------------|--------------------------|----------------------------|---|--|---|--|---|
| Austria | -377.22 | -507.65 | -1.04 | -26.05 | -2.44 | -28.49 | 8.00 | -18.05 |
| Belgium | -378.92 | -298.63 | 1.54 | -25.54 | -0.58 | -26.12 | 1.31 | -24.22 |
| Denmark | -188.01 | -266.24 | 18.37 | -12.72 | 0.71 | -12.01 | 43.07 | 30.35 |
| Finland | -204.37 | -162.88 | 23.87 | -11.74 | 1.37 | -10.37 | 10.47 | -1.27 |
| France | -1649.13 | -1205.50 | 14.47 | -83.06 | 29.58 | -53.48 | 11.72 | -71.34 |
| Germany | -2164.45 | -1937.47 | 18.61 | -205.97 | 5.90 | -200.06 | 140.89 | -65.07 |
| Ireland | -254.84 | -168.89 | -3.75 | -13.47 | -6.13 | -19.61 | -8.82 | -22.29 |
| Italy | -886.77 | -2063.00 | 1.39 | -40.39 | -8.09 | -48.48 | 19.35 | -21.04 |
| Luxembourg | -18.39 | -12.17 | 11.20 | -0.88 | 0.29 | -0.58 | 0.11 | -0.77 |
| Sweden | -229.51 | -400.16 | 39.63 | -23.15 | 0.98 | -22.16 | 21.33 | -1.82 |
| Switzerland | -193.86 | -129.64 | 4.19 | -7.01 | -0.43 | -7.43 | 34.46 | 27.45 |
| The Netherlands | -407.96 | -145.59 | 20.82 | -18.72 | 11.48 | -7.25 | 73.67 | 54.94 |
| United Kingdom | -1034.26 | -545.56 | 8.40 | -46.30 | 1.43 | -44.88 | -6.37 | -52.67 |
| Total | -7987.70 | -7843.38 | - | -514.99 | 34.06 | -480.93 | 349.20 | -165.79 |
| | | | Total Tg CO ₂ e | -17.51 | 1.16 | -16.35 | 11.87 | -5.64 |

Table 6 Nitrous oxide emissions modeling inputs and results. EF_{N₂O} = emissions factor, TAM = typical animal mass, N_{ex} = country- or region-specific N excretion rates. EF_{N₂O} uncertainty range is a factor of 2 for all but anaerobic lagoons and whole barns

| | EF _{N₂O} (kg N ₂ O- N kg N excreted ⁻¹) | TAM (kg hd ⁻¹) | N _{ex} (kg N 1000 kg TAM ⁻¹ d ⁻¹) | EF _{N₂O} and N _{ex} sources | N ₂ O emission rate | | Field- derived EF _{N₂O} (kg N ₂ O- N kg N excreted ⁻¹) |
|--------------------------------|---|-------------------------------|---|--|---|---|--|
| | | | | | Modeled (kg hd ⁻¹ yr ⁻¹) | Field (kg hd ⁻¹ yr ⁻¹) | |
| Anaerobic lagoon | 0 | 600 | 0.25 | Average of ID and TX* | 0 | 0.9 ± 0.5 (4) | 0.010 ± 0.005 |
| Slurry tanks and ponds | 0.005 | 600 | 0.44 | North America, with crust† | 0.8 | 0.3 ± 0.3 (3) | 0.005 ± 0.005 |
| Manure pile | 0.005 | 600 | 0.26 | ID* | 0.4 | 1.1 ± 0.7 (4) | 0.033 ± 0.020 |
| Corrals | 0.02 | 600 | 0.25 | Average of ID and TX* | 1.7 | 1.5 ± 0.8 (4) | 0.048 ± 0.026 |
| Barn floors and paved surfaces | 0.02 | 600 | 0.44 | North America† | 3.0 | 0.02 ± 0.01 (6) | 0.0001 ± 0.0001 |
| Whole barn (deep pit) | 0.002 | 600 | 0.48 | Western Europe† | 0.3 | 10 ± 6 (3) | 0.062 ± 0.038 |

*from (EPA, 2013b).

†from (IPCC, 2006).

Data were conspicuously lacking from India and China, which have the fastest growing dairy industries in the world (growing by 10.7 and 7.6 million hd, respectively, between 2000 and 2010; FAO, 2014).

Though data were not available on manure management practices in the two countries, estimated CH₄ emissions for each suggest that China is treating more manure in liquid form; for every million dairy milk

Table 7 Comparison of N₂O emissions from anaerobic lagoons and solid manure piles modeled for the United States using the current EF_{N₂O} (0 and 0.005 kg N₂O-N kg N excreted⁻¹, respectively) (EPA, 2013a) and the revised EF_{N₂O} (0.01 ± 0.005 and 0.033 ± 0.02 kg N₂O-N kg N excreted⁻¹, respectively) from Table 6. For all states, dairy cow TAM = 680 kg, heifer TAM = 407, and heifer N_{ex} = 69 kg N 1000 kg TAM⁻¹ d⁻¹

| State | Milk cows (hd) | Heifers (hd) | Manure in solid pile (%) | Manure in anaerobic lagoons (%) | N _{ex} dairy cows (kg N hd ⁻¹ yr ⁻¹) | EPA | | Revised | | EPA | | Revised | |
|---------------|----------------|--------------|--------------------------|---------------------------------|--|---|---|---|---|--|--|--|--|
| | | | | | | solid pile N ₂ O emissions (kg N ₂ O yr ⁻¹) | solid pile N ₂ O emissions (kg N ₂ O yr ⁻¹) | solid pile N ₂ O emissions (kg N ₂ O yr ⁻¹) | solid pile N ₂ O emissions (kg N ₂ O yr ⁻¹) | lagoons N ₂ O emissions (kg N ₂ O yr ⁻¹) | lagoons N ₂ O emissions (kg N ₂ O yr ⁻¹) | lagoons N ₂ O emissions (kg N ₂ O yr ⁻¹) | lagoons N ₂ O emissions (kg N ₂ O yr ⁻¹) |
| Alabama | 11 000 | 6000 | 7 | 16 | 130 | 627 | 4141 ± 2510 | 0 | 2869 ± 1434 | | | | |
| Alaska | 500 | 200 | 34 | 24 | 128 | 131 | 866 ± 525 | 0 | 185 ± 93 | | | | |
| Arizona | 185 000 | 60 000 | 9 | 61 | 159 | 15 336 | 101 217 ± 61 344 | 0 | 207 887 ± 103 944 | | | | |
| Arkansas | 12 000 | 4000 | 10 | 9 | 122 | 870 | 5745 ± 3482 | 0 | 1567 ± 783 | | | | |
| California | 1 750 000 | 760 000 | 9 | 59 | 158 | 148 050 | 977 127 ± 592 198 | 0 | 1 941 095 ± 970 547 | | | | |
| Colorado | 123 000 | 75 000 | 11 | 64 | 159 | 13 314 | 87 874 ± 53 257 | 0 | 154 930 ± 77 465 | | | | |
| Connecticut | 19 000 | 9500 | 16 | 13 | 145 | 2691 | 17 757 ± 10 762 | 0 | 4372 ± 2186 | | | | |
| Delaware | 5000 | 3000 | 19 | 10 | 143 | 852 | 5621 ± 3406 | 0 | 896 ± 448 | | | | |
| Florida | 114 000 | 30 000 | 7 | 43 | 149 | 6816 | 44 986 ± 27 265 | 0 | 83 741 ± 41 871 | | | | |
| Georgia | 78 000 | 23 000 | 9 | 23 | 147 | 5970 | 39 404 ± 23 881 | 0 | 30 515 ± 15 257 | | | | |
| Hawaii | 2000 | 1000 | 9 | 57 | 130 | 145 | 956 ± 580 | 0 | 1835 ± 918 | | | | |
| Idaho | 574 000 | 320 000 | 11 | 65 | 157 | 60 731 | 400 822 ± 242 922 | 0 | 717 725 ± 358 863 | | | | |
| Illinois | 98 000 | 46 000 | 39 | 16 | 146 | 33 772 | 222 897 ± 135 089 | 0 | 27 711 ± 13 855 | | | | |
| Indiana | 172 000 | 62 000 | 29 | 24 | 150 | 43 943 | 290 021 ± 175 770 | 0 | 72 733 ± 36 366 | | | | |
| Iowa | 210 000 | 130 000 | 34 | 20 | 152 | 67 738 | 447 069 ± 270 951 | 0 | 79 692 ± 39 846 | | | | |
| Kansas | 122 000 | 85 000 | 21 | 36 | 151 | 24 608 | 162 414 ± 98 432 | 0 | 84 371 ± 42 185 | | | | |
| Kentucky | 77 000 | 50 000 | 14 | 3 | 134 | 9262 | 61 132 ± 37 050 | 0 | 3970 ± 1985 | | | | |
| Louisiana | 19 000 | 5000 | 10 | 9 | 125 | 1379 | 9103 ± 5517 | 0 | 2483 ± 1241 | | | | |
| Maine | 32 000 | 17 000 | 20 | 10 | 144 | 5674 | 37 450 ± 22 697 | 0 | 5674 ± 2837 | | | | |
| Maryland | 53 000 | 28 000 | 22 | 8 | 144 | 10 330 | 68 179 ± 41 320 | 0 | 7513 ± 3756 | | | | |
| Massachusetts | 13 500 | 7500 | 22 | 8 | 138 | 2554 | 16 856 ± 10 216 | 0 | 1857 ± 929 | | | | |
| Michigan | 361 000 | 148 000 | 24 | 29 | 158 | 80 977 | 534 445 ± 323 906 | 0 | 195 693 ± 97 847 | | | | |
| Minnesota | 470 000 | 290 000 | 39 | 17 | 145 | 166 961 | 1 101 942 ± 667 844 | 0 | 145 556 ± 72 778 | | | | |
| Mississippi | 15 000 | 7000 | 10 | 12 | 135 | 1236 | 8160 ± 4946 | 0 | 2967 ± 1484 | | | | |
| Missouri | 95 000 | 50 000 | 42 | 11 | 131 | 32 560 | 214 898 ± 130 241 | 0 | 17 055 ± 8528 | | | | |
| Montana | 14 000 | 7000 | 19 | 42 | 150 | 2425 | 16 007 ± 9701 | 0 | 10 722 ± 5361 | | | | |
| Nebraska | 58 000 | 20 000 | 26 | 29 | 150 | 13 233 | 87 337 ± 52 932 | 0 | 29 520 ± 14 760 | | | | |
| Nevada | 28 000 | 10 000 | 10 | 65 | 157 | 2569 | 16 958 ± 10 277 | 0 | 33 402 ± 16 701 | | | | |
| New Hampshire | 15 000 | 7500 | 19 | 10 | 149 | 2583 | 17 050 ± 10 333 | 0 | 2719 ± 1360 | | | | |
| New Jersey | 7500 | 4000 | 25 | 6 | 138 | 1603 | 10 581 ± 6412 | 0 | 769 ± 385 | | | | |
| New Mexico | 322 000 | 140 000 | 9 | 61 | 164 | 28 173 | 185 944 ± 112 693 | 0 | 381 905 ± 190 953 | | | | |
| New York | 610 000 | 330 000 | 17 | 13 | 151 | 96 041 | 633 869 ± 384 163 | 0 | 146 886 ± 73 443 | | | | |

Table 7 (continued)

| State | Milk cows (hd) | Heifers (hd) | Manure in solid pile (%) | Manure in anaerobic lagoons (%) | N_{ex} dairy cows (kg $N\text{ hd}^{-1}\text{ yr}^{-1}$) | EPA solid pile N_2O emissions (kg $N_2O\text{ yr}^{-1}$) | Revised solid pile N_2O emissions (kg $N_2O\text{ yr}^{-1}$) | EPA lagoons N_2O emissions (kg $N_2O\text{ yr}^{-1}$) | Revised lagoons N_2O emissions (kg $N_2O\text{ yr}^{-1}$) |
|----------------|----------------|--------------|--------------------------|---------------------------------|---|---|---|--|--|
| North Carolina | 44 000 | 21 000 | 11 | 10 | 152 | 4440 | 29 306 ± 17 761 | 0 | 8073 ± 4037 |
| North Dakota | 20 000 | 10 000 | 38 | 15 | 142 | 6604 | 43 590 ± 26 418 | 0 | 5214 ± 2607 |
| Ohio | 270 000 | 125 000 | 38 | 15 | 145 | 89 967 | 593 780 ± 359 867 | 0 | 71 026 ± 35 513 |
| Oklahoma | 54 000 | 20 000 | 21 | 45 | 140 | 9409 | 62 100 ± 37 636 | 0 | 40 325 ± 20 162 |
| Oregon | 121 000 | 65 000 | 11 | 50 | 150 | 12 245 | 80 815 ± 48 979 | 0 | 111 315 ± 55 658 |
| Pennsylvania | 543 000 | 310 000 | 24 | 6 | 147 | 118 770 | 783 881 ± 475 079 | 0 | 59 385 ± 29 692 |
| Rhode Island | 1100 | 500 | 25 | 5 | 141 | 235 | 1549 ± 939 | 0 | 94 ± 47 |
| South Carolina | 16 000 | 7000 | 8 | 18 | 144 | 1108 | 7315 ± 4433 | 0 | 4988 ± 2494 |
| South Dakota | 90 000 | 25 000 | 24 | 31 | 150 | 18 635 | 122 989 ± 74 539 | 0 | 48 140 ± 24 070 |
| Tennessee | 50 000 | 35 000 | 12 | 4 | 140 | 5415 | 35 737 ± 21 659 | 0 | 3610 ± 1805 |
| Texas | 425 000 | 230 000 | 11 | 58 | 155 | 44 298 | 292 368 ± 177 193 | 0 | 467 144 ± 233 572 |
| Utah | 87 000 | 42 000 | 15 | 56 | 151 | 11 918 | 78 662 ± 47 674 | 0 | 88 991 ± 44 496 |
| Vermont | 135 000 | 61 000 | 17 | 13 | 145 | 20 068 | 132 448 ± 80 271 | 0 | 30 692 ± 15 346 |
| Virginia | 95 000 | 49 000 | 11 | 5 | 145 | 9285 | 61 282 ± 37 140 | 0 | 8441 ± 4220 |
| Washington | 252 000 | 122 000 | 11 | 56 | 160 | 26 658 | 175 941 ± 106 631 | 0 | 271 425 ± 135 712 |
| West Virginia | 10 000 | 5000 | 23 | 7 | 134 | 1900 | 12 543 ± 7602 | 0 | 1157 ± 578 |
| Wisconsin | 1 265 000 | 700 000 | 38 | 17 | 150 | 443 940 | 2 930 005 ± 1 775 761 | 0 | 397 210 ± 198 605 |
| Wyoming | 6000 | 5000 | 19 | 43 | 150 | 1123 | 7413 ± 4493 | 0 | 5084 ± 2542 |
| | | | | | Total kg $N_2O\text{ yr}^{-1}$ | 1 709 175 | 11 280 554 ± 6 836 699 | 0 | 6 023 128 ± 3 011 564 |
| | | | | | Total Tg $CO_2e\text{ yr}^{-1}$ | 0.51 | 3.36 ± 2.04 | 0 | 1.79 ± 0.90 |

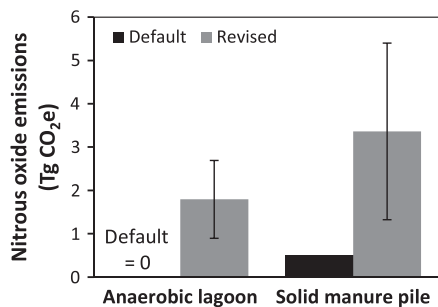


Fig. 5 Comparison of modeled N₂O emissions of solid manure piles and anaerobic lagoons calculated using current and revised EF_{N₂O} values.

cows gained between 2000 and 2010, India's CH₄ emissions from manure management increased by 5 Gg CH₄, whereas China's increased by 9 Gg CH₄ per million hd (FAO, 2014). Accurately modeling these emissions is critical for policy decisions toward GHG emission reduction.

The disagreement between field measurements and modeled values provides mechanistic support for discrepancies reported by airborne measurements and modeling. In a top-down approach combining aircraft and tower measurements with an atmospheric transport model, Miller *et al.* (2013) calculated total CH₄ emissions for the United States that were 1.5 times greater than the EPA bottom-up approach. Underestimation of emissions from fossil fuel extraction was responsible for a significant part of this discrepancy, but emissions from livestock enteric fermentation and manure management were calculated to be twice that of the EPA estimate (Miller *et al.*, 2013). A smaller scale analysis for the Los Angeles Basin found similar dairy CH₄ fluxes between top-down and bottom-up approaches (Peischl *et al.*, 2013).

Despite the uncertainties in emissions inventories described above, targets for GHG reduction can be identified. As shown by the European example, decreasing the number of cows can reduce GHG emissions by decreasing both enteric fermentation and manure production (Ripple *et al.*, 2013). While this is the trend in developed countries, developing nations have growing livestock populations which must be managed appropriately to be sustainable (Eisler *et al.*, 2014). The most effective GHG mitigation approach for manure management depends on how manure is handled and stored. Where liquid manure management systems are common, particularly anaerobic lagoons which were the highest total and per cow CH₄ source, they represent the greatest opportunity for GHG emissions reduction. Some estimates suggest that the total CH₄ and N₂O emissions

per head from anaerobic digesters are about 10% of the emissions from anaerobic lagoons (CARB, 2011a). The EPA (2011) estimated that adoption of anaerobic digesters by all US dairies for which this technology is feasible (those with liquid manure management systems and >500 hd, or approximately 2650 farms with 3 million hd) could reduce US CH₄ emissions by 41.25 Tg CO₂e yr⁻¹, or more than 85% of the total CH₄ emissions from dairy manure management. In addition, these anaerobic digesters would be capable of producing more than 6.8 million MWh yr⁻¹ (EPA, 2011). According to the GHG equivalency calculator at www.epa.gov, this would offset an additional 17.6 Tg CO₂e of CO₂ emissions from energy production.

Our results show significant disagreement between measured and modeled GHG emissions from dairies globally. Revised emission factors based on the field data led to greater estimated GHG emissions from the United States and Europe. More field data are needed to refine these models. To maximize the usefulness of field measurements, better reporting of herd characteristics (number of milk cows and heifers, average mass, milk production, dry matter, and N intake), dairy characteristics (manure handling practices and storage dimensions, climate parameters, available land for manure spreading, typical management schedule), and manure characteristics (amount handled by each storage method, volume, volatile solid content, C and N content, manure temperature) are needed. Emission rates must be reported with the data necessary to convert between units of per head, per area, and per kg milk production in addition to units specific to certain types of sources, such as per HPU for barns or per volume for liquid storage. Future research should focus on GHG emissions from several major dairy industries, particularly China (the fastest growing), India (the largest), and California (the largest in the United States) (FAO, 2006). Each region has unique issues related to climate, development, and legislation that complicate estimating GHG emissions without direct measurements. Furthermore, longer monitoring periods are needed to disentangle the effects of management and climate on emissions and enable more accurate estimates of annual averages.

Acknowledgements

This research was partially supported by grant #69-3S75-10-172 from the NRCS-USDA to the Environmental Defense Fund and W. Silver as part of the Conservation Innovation Program. Support was also generously provided by grants from the 11th Hour Project, the Lia Fund, and the Rathmann Family Foundation to W. Silver.

References

- Adviento-Borbe MA, Wheeler EF, Brown NE, Topper PA, Graves RE, Ishler VA, Varga GA (2010) Ammonia and greenhouse gas flux from manure in freestall barn with dairy cows on precision fed rations. *Transactions of the ASABE*, **53**, 1251–1266.
- Ahn HK, Mulbry W, White JW, Kondrad SL (2011) Pile mixing increases greenhouse gas emissions during composting of dairy manure. *Bioresource Technology*, **102**, 2904–2909.
- Amon B, Kryvoruchko V, Amon T, Zechmeister-Boltenstern S (2006) Methane, nitrous oxide and ammonia emissions during storage and after application of dairy cattle slurry and influence of slurry treatment. *Agriculture, Ecosystems & Environment*, **112**, 153–162.
- Bjorneberg DL, Leytem AB, Westermann DT, Griffiths PR, Shao L, Pollard MJ (2009) Measurement of atmospheric ammonia, methane, and nitrous oxide at a concentrated dairy production facility in southern Idaho using open-path FTIR spectrometry. *Transactions of the ASABE*, **52**, 1749–1756.
- Bogner J, Spokas K, Burton E, Sweeney R, Corona V (1995) Landfills as atmospheric methane sources and sinks. *Chemosphere*, **31**, 4119–4130.
- Borhan MS, Capareda S, Mukhtar S, Faulkner WB, McGee R, Parnell CB (2011a) Determining seasonal greenhouse gas emissions from ground-level area sources in a dairy operation in central Texas. *Journal of the Air & Waste Management Association*, **61**, 786–795.
- Borhan MS, Capareda SC, Mukhtar S, Faulkner WB, McGee R, Parnell CB (2011b) Greenhouse gas emissions from ground level area sources in dairy and cattle feed-yard operations. *Atmosphere*, **2**, 303–329.
- Borhan S, Mukhtar S, Capareda S, Rahman S (2012) Greenhouse gas emissions from housing and manure management systems at confined livestock operations. In: *Waste Management - An Integrated Vision* (ed Rebellon LFM). InTech, Available at: <http://www.intechopen.com/books/waste-management-an-integrated-vision/greenhouse-gas-emissions-from-housing-and-manure-management-systems-at-confined-livestock-operations> (accessed 12 June 2013). ISBN: 978-953-51-0795-8.
- Brown HA, Wagner-Riddle C, Thurtell GW (2002) Nitrous oxide flux from a solid dairy manure pile measured using a micrometeorological mass balance method. *Nutrient Cycling in Agroecosystems*, **62**, 53–60.
- CARB (2011a) *California Greenhouse Gas Emissions Inventory: 2000-2009*. California Air Resources Board, Sacramento, CA.
- CARB (2011b) *Annex 3B. Manure Management (IPCC 3A2) to the Technical Support Document for the 2000-2009 California's Greenhouse Gas Emissions Inventory*. California Air Resources Board, Sacramento, CA.
- Chadwick D, Sommer S, Thorman R, Fangueiro D, Cardenas L, Amon B, Misselbrook T (2011) Manure management: implications for greenhouse gas emissions. *Animal Feed Science and Technology*, **166–167**, 514–531.
- Chung ML, Shilton AN, Guieysse B, Pratt C (2013) Questioning the accuracy of greenhouse gas accounting from agricultural waste: a case study. *Journal of Environment Quality*, **42**, 654–659.
- Conrad R (2007) Microbial ecology of methanogens and methanotrophs. *Advances in Agronomy*, **96**, 1–63.
- Craggs R, Park J, Heubeck S (2008) Methane emissions from anaerobic ponds on a piggery and a dairy farm in New Zealand. *Australian Journal of Experimental Agriculture*, **48**, 142–146.
- Davidson EA (2009) The contribution of manure and fertilizer nitrogen to atmospheric nitrous oxide since 1860. *Nature Geoscience*, **2**, 659–662.
- Eisler MC, Lee MRF, Tarlton JF *et al.* (2014) Steps to sustainable livestock. *Nature*, **507**, 32–34.
- Ellis S, Webb J, Misselbrook T, Chadwick D (2001) Emission of ammonia (NH₃), nitrous oxide (N₂O) and methane (CH₄) from a dairy hardstanding in the UK. *Nutrient Cycling in Agroecosystems*, **60**, 115–122.
- Engström P, Dalsgaard T, Hulth S, Aller RC (2005) Anaerobic ammonium oxidation by nitrite (anammox): implications for N₂ production in coastal marine sediments. *Geochimica et Cosmochimica Acta*, **69**, 2057–2065.
- EPA (2011) *Market Opportunities for Biogas Recovery Systems at U.S. Livestock Facilities*. U.S. Environmental Protection Agency, Washington, DC.
- EPA (2012) *Summary Report: Global Anthropogenic Non-CO₂ Greenhouse Gas Emissions: 1990–2030*. Office of Atmospheric Programs, Climate Change Division, US Environmental Protection Agency, Washington, DC.
- EPA (2013a) *Inventory of US Greenhouse Gas Emissions and Sinks: 1990–2009*. US Environmental Protection Agency, Washington, DC.
- EPA (2013b) *Inventory of US Greenhouse Gas Emissions and Sinks: 1990–2009, Annexes*. US Environmental Protection Agency, Washington, DC.
- FAO, Steinfeld H, Gerber P, Wassenaar T, Castel V, Rosales M, De Haan C (2006) *Livestock's Long Shadow: Environmental Issues and Options*. Food and Agriculture Organization of the United Nations, Rome.
- FAO (2014) *FAOSTAT, Food and Agriculture Organization of the United Nations*. FAO Statistics Division, Rome, Italy. faostat3.fao.org.
- Gao Z, Yuan H, Ma W, Liu X, Desjardins RL (2011) Methane emissions from a dairy feedlot during the fall and winter seasons in Northern China. *Environmental Pollution*, **159**, 1183–1189.
- Goodland R (2014) A fresh look at livestock greenhouse gas emissions and mitigation potential in Europe. *Global Change Biology*, **20**, 2042–2044.
- Gupta PK, Jha AK, Koul S *et al.* (2007) Methane and nitrous oxide emission from bovine manure management practices in India. *Environmental Pollution (Barking, Essex: 1987)*, **146**, 219–224.
- Harter T, Davis H, Mathews MC, Meyer RD (2002) Shallow groundwater quality on dairy farms with irrigated forage crops. *Journal of Contaminant Hydrology*, **55**, 287–315.
- Hensen A, Groot TT, van den Bulk WCM, Vermeulen AT, Olesen JE, Schelde K (2006) Dairy farm CH₄ and N₂O emissions, from one square metre to the full farm scale. *Agriculture, Ecosystems & Environment*, **112**, 146–152.
- Husted S (1994) Seasonal Variation in Methane Emission from Stored Slurry and Solid Manures. *Journal of Environmental Quality*, **23**, 585–592.
- IPCC (1996) Chapter 2: Radiative forcing of climate change. In: *Climate Change 1995: The Science of Climate Change* (eds Joughton JT, Meira Filho LG, Callander BA, Harris N, Kattenberg A, Maskell K), pp. 65–132. Cambridge University Press, Cambridge, UK.
- IPCC (2006) Chapter 10: Emissions from livestock and manure management. In: *IPCC Guidelines for National Greenhouse Gas Inventories, Prepared by the National Greenhouse Gas Inventories Programme* (eds Eggleston S, Buendia L, Miwa K, Ngara T, Tanabe K), pp. 10.1–10.87. Cambridge University Press, Cambridge, United Kingdom.
- Jungbluth T, Hartung E, Brose G (2001) Greenhouse gas emissions from animal houses and manure stores. *Nutrient Cycling in Agroecosystems*, **60**, 133–145.
- Kaharabata SK, Schuepp PH, Desjardins RL (1998) Methane emissions from above-ground open manure slurry tanks. *Global Biogeochemical Cycles*, **12**, 545–554.
- Kaharabata SK, Schuepp PH, Desjardins RL (2000) Estimating Methane Emissions from Dairy Cattle Housed in a Barn and Feedlot Using an Atmospheric Tracer. *Environmental Science & Technology*, **34**, 3296–3302.
- Khan RZ, Muller C, Sommer SG (1997) Micrometeorological mass balance technique for measuring CH₄ emission from stored cattle slurry. *Biology and Fertility of Soils*, **24**, 442–444.
- Kinsman R, Sauer FD, Jackson HA, Wolynetz MS (1995) Methane and carbon dioxide emissions from dairy cows in full lactation monitored over a six-month period. *Journal of Dairy Science*, **78**, 2760–2766.
- Leytem AB, Dungan RS, Bjorneberg DL, Koehn AC (2011) Emissions of ammonia, methane, carbon dioxide, and nitrous oxide from dairy cattle housing and manure management systems. *Journal of Environmental Quality*, **40**, 1383–1394.
- Leytem AB, Dungan RS, Bjorneberg DL, Koehn AC (2013) Greenhouse gas and ammonia emissions from an open-freestall dairy in southern Idaho. *Journal of Environmental Quality*, **42**, 10–20.
- Luo J, Sagar S (2008) Nitrous oxide and methane emissions from a dairy farm stand-off pad. *Australian Journal of Experimental Agriculture*, **48**, 179.
- Maeda K, Toyoda S, Shimojima R, Osada T, Hanajima D, Morioka R, Yoshida N (2010) Source of nitrous oxide emissions during the cow manure composting process as revealed by isotopomer analysis of and amoA abundance in betaproteobacterial ammonia-oxidizing bacteria. *Applied and Environmental Microbiology*, **76**, 1555–1562.
- Mangino J, Bartram D, Brazy A (2002) Development of a methane conversion factor to estimate emissions from animal waste lagoons. In: *11th International Emission Inventory Conference*. U.S. Environmental Protection Agency, Washington, DC. Available at: <http://www.epa.gov/ttnchie1/conference/ei11/ammonia/mangino.pdf> (accessed 15 October 2012).
- Marik T, Levin I (1996) A new tracer experiment to estimate the methane emissions from a dairy cow shed using sulfur hexafluoride (SF₆). *Global Biogeochemical Cycles*, **10**, 413–418.
- Massé DI, Talbot G, Gilbert Y (2011) On farm biogas production: a method to reduce GHG emissions and develop more sustainable livestock operations. *Animal Feed Science and Technology*, **166–167**, 436–445.
- McCandlish AC (1920) Environment and breeding as factors influencing milk production. *The Journal of Heredity*, **11**, 204–214.
- McGinn SM, Beauchemin KA (2012) Dairy farm methane emissions using a dispersion model. *Journal of Environmental Quality*, **41**, 73–79.
- Miller SM, Wofsy SC, Michalak AM *et al.* (2013) Anthropogenic emissions of methane in the United States. *Proceedings of the National Academy of Sciences of the United States of America*, **110**, 20018–20022.

- Misselbrook T, Webb J, Chadwick D, Ellis S, Pain B (2001) Gaseous emissions from outdoor concrete yards used by livestock. *Atmospheric Environment*, **35**, 5331–5338.
- Monteny CJ, Groenestein CM, Hilhorst MA (2001) Interactions and coupling between emissions of methane and nitrous oxide from animal husbandry. *Nutrient Cycling in Agroecosystems*, **60**, 123–132.
- Mulder A, Graaf AA, Robertson LA, Kuenen JG (1995) Anaerobic ammonium oxidation discovered in a denitrifying fluidized bed reactor. *FEMS Microbiology Ecology*, **16**, 177–184.
- Myhre G, Shindell D, Breon F-M *et al.* (2013) Anthropogenic and natural radiative forcing. In: *Climate Change 2013: The Physical Science Basis. Contribution of Working Group 1 to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change* (eds Stocker TF, Qin D, Plattner G-K, Tignor M, Allen SK, Boschung J, Nauels A, Xia Y, Bex V, Midgley PM), Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.
- Ngwabie NM, Jeppsson K-H, Nimmermark S, Swensson C, Gustafsson G (2009) Multi-location measurements of greenhouse gases and emission rates of methane and ammonia from a naturally-ventilated barn for dairy cows. *Biosystems Engineering*, **103**, 68–77.
- Ngwabie NM, Jeppsson K-H, Gustafsson G, Nimmermark S (2011) Effects of animal activity and air temperature on methane and ammonia emissions from a naturally ventilated building for dairy cows. *Atmospheric Environment*, **45**, 6760–6768.
- O'Brien D, Shalloo L, Patton J, Buckley F, Grainger C, Wallace M (2012) Evaluation of the effect of accounting method, IPCC v. LCA, on grass-based and confinement dairy systems' greenhouse gas emissions. *Animal*, **6**, 1512–1527.
- Osada T, Sommer SG, Dahl P, Rom HB (2001) Gaseous emission and changes in nutrient composition during deep litter composting. *Acta Agriculturae Scandinavica, Section B - Soil & Plant Science*, **51**, 137–142.
- Parkin TB, Kaspar TC (2004) Temporal variability of soil carbon dioxide flux: effect of sampling frequency on cumulative carbon loss estimation. *Soil Science Society of America Journal*, **68**, 1234–1241.
- Peischl J, Ryerson TB, Brioude J *et al.* (2013) Quantifying sources of methane using light alkanes in the Los Angeles basin, California. *Journal of Geophysical Research: Atmospheres*, **118**, 4974–4990.
- Petersen SO, Sommer SG (2011) Ammonia and nitrous oxide interactions: roles of manure organic matter management. *Animal Feed Science and Technology*, **166–167**, 503–513.
- Petersen SO, Amon B, Gattinger A (2005) Methane oxidation in slurry storage surface crusts. *Journal of Environmental Quality*, **34**, 455–461.
- Place SE, Mitloehner FM (2010) Invited review: contemporary environmental issues: a review of the dairy industry's role in climate change and air quality and the potential of mitigation through improved production efficiency. *Journal of Dairy Science*, **93**, 3407–3416.
- Ripple WJ, Smith P, Haberl H, Montzka SA, McAlpine C, Boucher DH (2013) Ruminants, climate change and climate policy. *Nature Climate Change*, **4**, 2–5.
- RIVM, Coenen PWHG, van der Maas CWM *et al.* (2013) *Greenhouse gas Emission in The Netherlands*. National Institute for Public Health and the Environment (RIVM), Bilthoven, The Netherlands.
- Safley LM, Westerman PW (1988) Biogas production from anaerobic lagoons. *Biological Wastes*, **23**, 181–193.
- Safley LM, Westerman PW (1992) Performance of a dairy manure anaerobic lagoon. *Bioresource Technology*, **42**, 43–52.
- Samer M, Ammon C, Loebin C, Fiedler M, Berg W, Sanftleben P, Brunsch R (2012) Moisture balance and tracer gas technique for ventilation rates measurement and greenhouse gases and ammonia emissions quantification in naturally ventilated buildings. *Building and Environment*, **50**, 10–20.
- Sedorovich DM, Rotz CA, Richard TL (2007) Greenhouse gas emissions from dairy farms. In: 2007 ASABE Annual International Meeting, Vol. 0300, p. Paper No. 074096. American Society of Agricultural and Biological Engineers.
- Sneath RW, Beline F, Hilhorst MA, Peu P (2006) Monitoring GHG from manure stores on organic and conventional dairy farms. *Agriculture, Ecosystems & Environment*, **112**, 122–128.
- Snell HGJ, Seipelt F, Van den Weghe HFA (2003) Ventilation rates and gaseous emissions from naturally ventilated dairy houses. *Biosystems Engineering*, **86**, 67–73.
- Sommer SG, McGinn SM, Hao X, Larney FJ (2004) Techniques for measuring gas emissions from a composting stockpile of cattle manure. *Atmospheric Environment*, **38**, 4643–4652.
- Tauseef SM, Premalatha M, Abbasi T, Abbasi SA (2013) Methane capture from livestock manure. *Journal of Environmental Management*, **117**, 187–207.
- Todd RW, Cole NA, Rhoades MB, Parker DB, Casey KD (2008) Daily, monthly, seasonal, and annual ammonia emissions from Southern High Plains cattle feedyards. *Journal of Environmental Quality*, **40**, 1090–1095.
- Todd RW, Cole NA, Casey KD, Hagevoort R, Auvermann BW (2011) Methane emissions from southern High Plains dairy wastewater lagoons in the summer. *Animal Feed Science and Technology*, **166–167**, 575–580.
- UNFCCC (2014) 2013 annex 1 party GHG national inventory submissions. 2013 Annex 1 Party GHG National Inventory Submissions.
- USDA (2011) *U.S. Agriculture and Forestry Greenhouse Gas Inventory: 1990–2008. Technical Bulletin No. 1930*. Climate Change Program Office, Office of the Chief Economist, U.S. Department of Agriculture, Washington, DC.
- VanderZaag AC, Wagner-Riddle C, Park K-H, Gordon RJ (2011) Methane emissions from stored liquid dairy manure in a cold climate. *Animal Feed Science and Technology*, **166–167**, 581–589.
- van Vliet LJP, Patni NK, Matin MA (2004) GHG emissions from manure in a naturally ventilated, freestall dairy barn, comparing sand and sawdust bedding. In: *ASAE/CSAE Meeting Paper No. 044161*, Vol. 0300. ASAE, St. Joseph, MI.
- Wu W, Zhang G, Kai P (2012) Ammonia and methane emissions from two naturally ventilated dairy cattle buildings and the influence of climatic factors on ammonia emissions. *Atmospheric Environment*, **61**, 232–243.
- Yamulki S (2006) Effect of straw addition on nitrous oxide and methane emissions from stored farmyard manures. *Agriculture, Ecosystems & Environment*, **112**, 140–145.
- Yang WH, Weber KA, Silver WL (2012) Nitrogen loss from soil through anaerobic ammonium oxidation coupled to iron reduction. *Nature Geoscience*, **5**, 538–541.
- Zhang G, Strøm JS, Li B, Rom HB, Morsing S, Dahl P, Wang C (2005) Emission of ammonia and other contaminant gases from naturally ventilated dairy cattle buildings. *Biosystems Engineering*, **92**, 355–364.
- Zhu Z, Dong H, Zhou Z (2012) Ammonia and greenhouse gas emissions from a dairy cattle barn with a daily manure collection system. *Transactions of the ASABE*, **55**, 1959–1964.
- Zimmerman CA, Rakes AH, Jaquette RD, Hopkins BA, Croom WJ (1991) Effects of protein level and forage source on milk production and composition in early lactation dairy cows. *Journal of Dairy Science*, **74**, 980–990.

Supporting Information

Additional Supporting Information may be found in the online version of this article:

Table S1. Summary of the measurement techniques, locations, and climate data for each study.

Table S2. Anaerobic lagoon study characteristics and GHG emissions.

Table S3. Slurry tank and pond study characteristics and GHG emissions.

Table S4. Corral study characteristics and GHG emissions.

Table S5. Solid manure pile study characteristics and GHG emissions.

Table S6. Whole barn and barn floor study characteristics and GHG emissions.

Table S7. Hardstanding study characteristics and GHG emissions.

Table S8. Whole dairy study characteristics and GHG emissions.