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RESEARCH REVIEW

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

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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 CH_4 hd^{-1} yr^{-1}), more than three times that from enteric fermentation (~120 kg CH_4 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 N_2O hd^{-1} yr^{-1}, respectively). Nitrous oxide emissions from anaerobic lagoons (0.9 ± 0.5 kg N_2O hd^{-1} yr^{-1}) and barns (10 ± 6 kg N_2O 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 CH_4 emissions for Europe and increased N_2O 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

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Introduction

Animal agricultural currently accounts for 20% of non-CO_2 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 CH_4 emissions from dairies were from manure management (USDA, 2011), whereas in California, the state with the greatest dairy production, 54% of dairy CH_4 was estimated to come from manure management (CARB, 2011a). Manure management can also be an important source of nitrous oxide (N_2O) emissions, accounting for an estimated 5% of global (EPA, 2012) and US (EPA, 2013a) N_2O emissions. Modeling estimates suggested that N_2O emissions from manure management globally played a dominant role in the atmospheric increase in N_2O over the last 140 years (Davidson, 2009).

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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

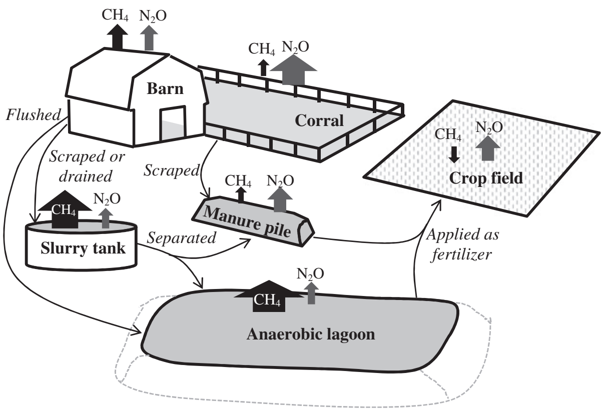

Fig. 1 Sources of N_2O and CH_4 on dairies. Thin arrows indicate movement of manure between locations. Thick arrows indicate relative emission rate. Hardstandings are not shown but have negligible emissions.

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 & Saggar, 2008). The sheeting allows the leachate from the pad to be collected and treated in liquid storage systems. Hardstandings 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 kg hd^{-1} d^{-1} for North American studies and 117 kg hd^{-1} d^{-1} for European and modern, high-producing Chinese dairies (rather than the default of 68 kg hd^{-1} 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 (kg trace gas hd^{-1} yr^{-1}) and per unit area per time (kg trace gas m^{-2} 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 *et al.* (2011a)	×			×	×		×	
Borhan *et al.* (2011b)	×				×		×	
Bjorneberg *et al.* (2009)	×				×			
Leytem *et al.* (2011)	×			×	×			
Leytem *et al.* (2013)	×						×	
Craggs *et al.* (2008)	×							
Safley and Westerman (1988)	×							
Safley and Westerman (1992)	×							
Todd *et al.* (2011)	×							
Husted (1994)		×	×					
Sneath *et al.* (2006)*		×	×					
Hensen *et al.* (2006)		×						×
Kaharabata *et al.* (1998)		×						
Khan *et al.* (1997)		×						
VanderZaag *et al.* (2011)		×						
Ahn *et al.* (2011)			×					
Amon *et al.* (2006)			×					
Brown *et al.* (2002)			×					
Gupta *et al.* (2007)			×					
Osada *et al.* (2001)			×					
Sommer *et al.* (2004)			×					
Kaharabata *et al.* (2000)					×			
Luo and Saggar (2008)					×			
Ellis *et al.* (2001)						×		
Gao *et al.* (2011)						×		
Misselbrook *et al.* (2001)						×		
Adviento-Borbe *et al.* (2010)							×	
Kinsman *et al.* (1995)							×	
Marik and Levin (1996)							×	
Ngwabie *et al.* (2009)							×	
Ngwabie *et al.* (2011)							×	
Samer *et al.* (2012)							×	
Snell *et al.* (2003)							×	
van Vliet *et al.* (2004)							×	
Wu *et al.* (2012)							×	
Zhang *et al.* (2005)							×	
Zhu *et al.* (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, N_2O and CH_4 emission rates were converted to 100-year CO_2e 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 N_2O and CH_4 of 310 and 21 (IPCC, 1996), respectively, those emission rates were

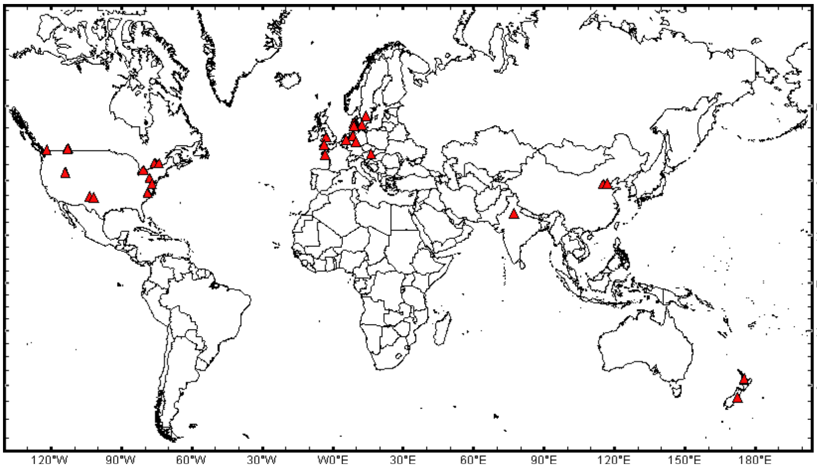

Fig. 2 Geographic distribution of sampling sites, marked as triangles (created in GeoMapApp v. 3.3.8, http://www.geomapapp.org/).

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 CO_2e hd^{-1} yr^{-1} and 3.5 ± 1.7 Mg CO_2e 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 CO_2e m^{-2} yr^{-1} and 827 ± 320 kg CO_2e m^{-2} yr^{-1}, respectively (Table 2b). These rates were high; for comparison, the highest landfill CH_4 emissions rates reported in Bogner *et al.* (1995) were 248 kg CO_2e m^{-2} yr^{-1}, less than half those from liquid manure systems. Barn floors had the lowest GWP (38 ± 7 kg CO_2e 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 CH_4, with anaerobic lagoons and slurry stores emitting 368 ± 193 kg CH_4 hd^{-1} yr^{-1} and 101 ± 47 kg CH_4 hd^{-1} yr^{-1}, respectively (Table 2a). Barns were the next largest source with 33 ± 19 kg CH_4 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 N_2O emissions by nearly an order of magnitude, with 10.3 ± 6.2 kg N_2O hd^{-1} yr^{-1} (Table 2a), although field data were highly variable (Table S6). Corrals and solid manure piles were the next largest N_2O source with 1.5 ± 0.8 kg N_2O hd^{-1} yr^{-1} and 1.1 ± 0.7 kg N_2O 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 N_2O hd^{-1} yr^{-1} and 0.3 ± 0.3 kg N_2O hd^{-1} yr^{-1}, respectively. The relatively large net N_2O flux from liquid manure storage was surprising given the predominantly anaerobic conditions typical of unaerated systems. Nitrogen in liquid manure is mostly in the form of ammonium (NH_4^+) and organic N (Harter *et al.*, 2002), and though anaerobic lagoons are generally anaerobic, aerobic conditions which could promote denitrification exist at inlets. Other N_2O formation reactions are also feasible, such as denitrification of nitrate (NO_3^-) produced through annamox [anaerobic NH_4^+ oxidation, (Mulder *et al.*, 1995; Maeda *et al.*, 2010)], Feammox [anaerobic NH_4^+ oxidation coupled to Fe reduction, (Yang *et al.*, 2012)], or Mnammox [anaerobic NH_4^+ oxidation coupled to Mn reduction, (Engström *et al.*, 2005)]. Hardstandings and barn floors, surfaces which were scraped or flushed frequently, had CH_4 and N_2O 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 CH_4 emissions are positively correlated with MAT (Mangino *et al.*, 2002; IPCC, 2006). Individual field studies observed greater CH_4 emissions in summer and/or with warmer sampling temperatures for manure piles, barns, and whole dairies; however, there was no significant correlation between CH_4 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 N_2O and CH_4 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_2e hd^{-1} yr^{-1})		
	Mean* ± SE (*n*) Range			Mean* ± SE (*n*)		
	CH_4	N_2O	CO_2	From CH_4	From N_2O	Total (CH_4+N_2O)
(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

	Emission rate (kg m^{-2} yr^{-1})			Emission global warming potential† (kg CO_2e m^{-2} yr^{-1})
	Mean* ± SE (*n*) Range			Mean* ± SE (*n*)
	CH_4	N_2O	CO_2	Total (CH_4+N_2O)
(b)				
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 CH_4 = 34 g CO_2e and 1 g N_2O = 298 g CO_2e. CO_2 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 CH_4 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 CH_4 emissions within or across studies.

Methane and N_2O 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 N_2O fluxes were derived from denitrification, which requires the same general environmental conditions as methanogenesis (warm temperatures, abundant labile C, anaerobic conditions). Corral CH_4 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 N_2O 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 CH_4 nor N_2O emissions from corrals were correlated with temperature and/or precipitation. Leytem *et al.* (2011) measured higher N_2O 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 CH_4 and N_2O 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 CH_4 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 CH_4 fluxes. Borhan *et al.* (2011) measured greater CH_4 and N_2O 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 CH_4 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 CH_4 emissions (EF_{CH4}, g CH_4 hd^{-1} yr^{-1}) based on the volatile solid production by the cows (VS, kg VS hd^{-1} yr^{-1}), a CH_4 conversion factor (MCF, %) for the manure management practice, and the maximum possible CH_4 production rate from the volatile solids in the manure (B_o, m^3 CH_4 kg VS^{-1}):

$$EF_{CH4} = VS \times MCF/100 \times B_0 \times 662 \text{ g } CH_4 \text{ m}^{-3} CH_4 \quad (1)$$

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 N_2O 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 N_2O-N kg N excreted^{-1}) (equations 10.31, 10.32, and 10.33 in IPCC, 2006). EF_{N2O} can be converted into N_2O emission rates equivalent to those measured here (N_2O_D, g N_2O 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}):

$$N_2O_D = EF_{N2O} \times 44/28 \times TAM \times N_{ex} \quad (2)$$

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 N_2O emissions, derived from the oxidation of gaseous emissions such as ammonia (NH_3) 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 N_2O 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 CH_4 emissions

The means of the field-measured CH_4 emissions from slurry tanks and barns (deep pit storage) were three times larger than modeled emissions, while the measured CH_4 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 CH_4 emissions from the remaining sources (anaerobic lagoons and hardstandings) were within the standard error of the field means or were negligible. Modeled whole dairy CH_4 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 CH_4 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 CH_4 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, B_o = the maximum possible CH_4 production rate from the volatile solids in the manure

	VS (kg VS hd^{-1} yr^{-1})	VS data source	MCF (%)	MCF data source	B_o (m^3 CH_4 kg VS^{-1})	B_o data source	CH_4 emission rate Modeled (kg hd^{-1} yr^{-1})	CH_4 emission rate 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=11, 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).

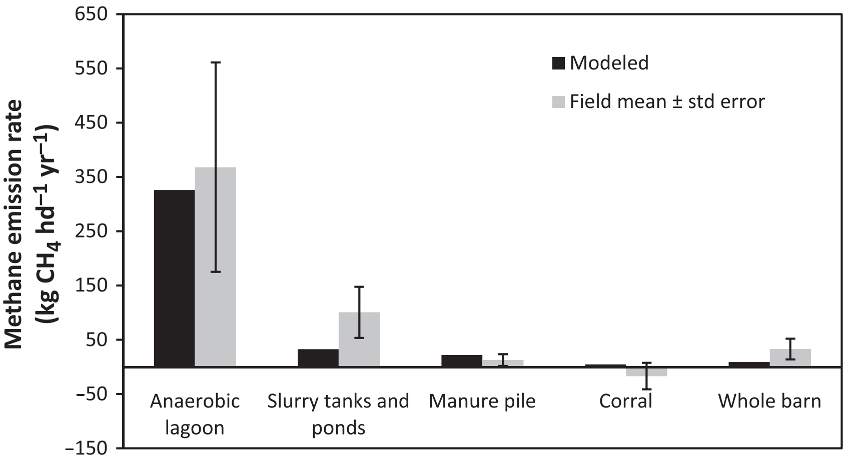

Fig. 3 Comparison of modeled CH_4 emissions and field measurement means and standard errors for the largest CH_4 sources.

MCFs (8.4 ± 4.6 Tg CO_2e yr^{-1} vs. 15.2 Tg CO_2e yr^{-1}, respectively, Table 4, Fig. 4). However, the revised slurry pond MCF increased CH_4 emissions from slurry for most countries, with total emissions of 25.9 ± 12.2 Tg CO_2e yr^{-1}, a gain of 10.7 Tg CO_2e 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 CH_4 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

								Liquid manure emissions (Gg CH_4 yr^{-1})		
	Dairy cows (1000 hd)	VS (kg DM hd^{-1} d^{-1})	Bo (m3 CH4 kg^{-1} VS)	Liquid fraction (%)	Liquid MCF (%)	Revised MCF (deep pit) (%)	Revised MCF (slurry) (%)	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 CO2e)	15.2	8.4 ± 4.6	25.9 ± 12.2
							Top 5 liquid (Tg CO2e)	6.6	5.4 ± 2.9	16.7 ± 7.8

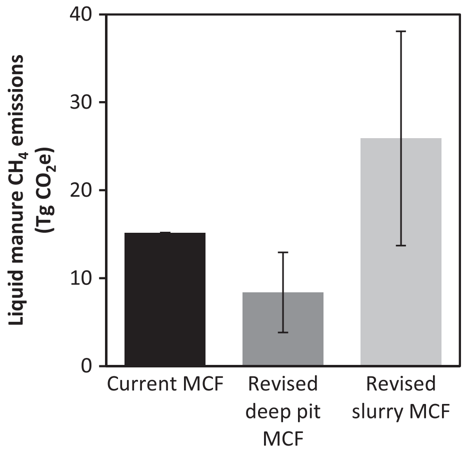

Fig. 4 Comparison of modeled CH_4 emissions for slurry using the current slurry MCF used by the European Union and the revised MCFs for deep pit storage and slurry storage calculated in this study (Table 4).

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 CH_4 (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 CH_4 production; the current estimates suggest a total net decrease (combined change in enteric and manure management emissions) of 480 Gg CH_4 (Table 5). Using the revised slurry MCF for the 2011 estimates gives a smaller total net decrease of 166 Gg CH_4, with some countries (Denmark, Switzerland, and the Netherlands) having net increases of 30–50 Gg CH_4 rather than decreases (Table 5).

Measured vs. modeled N_2O emissions

Modeled N_2O 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 N_2O emissions from anaerobic lagoons, whereas the revised EF_{N2O} gave 1.79 ± 0.90 Tg $CO_2e\ yr^{-1}$ (Fig. 5). Nitrous oxide emissions from solid manure piles also increased from 0.51 Tg $CO_2e\ yr^{-1}$ to 3.36 ± 2.04 Tg $CO_2e\ yr^{-1}$ using the revised EF_{N2O} (Fig. 5). Combined, the revised values increased manure management N_2O emissions in the United States by more than 4.5 Tg $CO_2e\ yr^{-1}$, 25% of the 2011 estimate of 17.3 Tg CO_2e (EPA, 2013a).

Whole barn N_2O 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 N_2O from dairies (2–3 times more kg N_2O $hd^{-1}\ yr^{-1}$ than corrals or solid piles). No standard model has been established for calculating N_2O 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 $N_2O\ hd^{-1}\ yr^{-1}$ (the approximate mean of the measurements by Zhang *et al.* (2005), which was the lowest of the three studies that measured N_2O), then barns emitted 1.64 Tg CO_2e, on the same scale as the revised N_2O emissions from anaerobic lagoons in the United States (Table 7). Using the field measurement mean of 10 kg $N_2O\ hd^{-1}\ yr^{-1}$ for the calculation increased barn emissions by an order of magnitude (to 16.4 Tg CO_2e), equivalent to the warming potential of slurry system-derived CH_4 in Europe (Table 4). More data are needed to assess if barns are actually such large sources of N_2O.

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_2e	−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 N2O-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_2O emission rate: Modeled (kg hd^{-1} yr^{-1})	N_2O emission rate: Field (kg hd^{-1} yr^{-1})	Field-derived EF_{N2O} (kg N_2O-N kg N excreted^{-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 CH_4 emissions for each suggest that China is treating more manure in liquid form; for every million dairy milk

Table 7 Comparison of N_2O emissions from anaerobic lagoons and solid manure piles modeled for the United States using the current EF_{N2O} (0 and 0.005 kg N_2O-N kg N excreted^{-1}, respectively) (EPA, 2013a) and the revised EF_{N2O} (0.01 ± 0.005 and 0.033 ± 0.02 kg N_2O-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 solid pile N_2O emissions (kg N_2O yr^{-1})	Revised solid pile N_2O emissions (kg N_2O yr^{-1})	EPA lagoons N_2O emissions (kg N_2O yr^{-1})	Revised lagoons N_2O emissions (kg N_2O 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 hd^{-1} yr^{-1})	EPA solid pile N_2O emissions (kg N_2O yr^{-1})	Revised solid pile N_2O emissions (kg N_2O yr^{-1})	EPA lagoons N_2O emissions (kg N_2O yr^{-1})	Revised lagoons N_2O emissions (kg N_2O 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 N2O yr^{-1}	1 709 175	11 280 554 ± 6 836 699	0	6 023 128 ± 3 011 564
					Total Tg CO_2e yr^{-1}	0.51	3.36 ± 2.04	0	1.79 ± 0.90

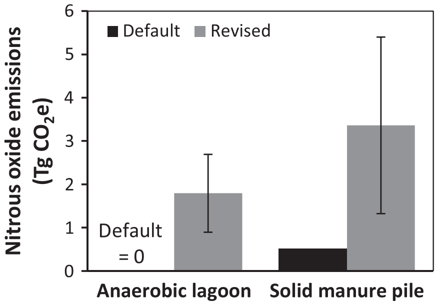

Fig. 5 Comparison of modeled N_2O emissions of solid manure piles and anaerobic lagoons calculated using current and revised EF_{N2O} values.

cows gained between 2000 and 2010, India's CH_4 emissions from manure management increased by 5 Gg CH_4, whereas China's increased by 9 Gg CH_4 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 CH_4 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 CH_4 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 CH_4 source, they represent the greatest opportunity for GHG emissions reduction. Some estimates suggest that the total CH_4 and N_2O 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 CH_4 emissions by 41.25 Tg CO_2e yr^{-1}, or more than 85% of the total CH_4 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 CO_2e of CO_2 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.

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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.