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4

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

6

7 Running head: GHG emissions from dairy manure management

8

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15

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

17

18 Research Review

19 **Abstract**

20 Livestock manure management accounts for almost 10% of greenhouse gas emissions from
21 agriculture globally, and contributes an equal proportion to the US methane emission inventory.
22 Current emissions inventories use emissions factors determined from small-scale laboratory
23 experiments that have not been compared to field-scale measurements. We compiled published
24 data on field-scale measurements of greenhouse gas emissions from working and research dairies
25 and compared these to rates predicted by the IPCC Tier 2 modeling approach. Anaerobic
26 lagoons were the largest source of methane ($368 \pm 193 \text{ kg CH}_4 \text{ hd}^{-1} \text{ y}^{-1}$), more than three times
27 that from enteric fermentation ($\sim 100 \text{ kg CH}_4 \text{ hd}^{-1} \text{ y}^{-1}$). Corrals and solid manure piles were large
28 sources of nitrous oxide (1.5 ± 0.8 and $1.1 \pm 0.7 \text{ kg N}_2\text{O hd}^{-1} \text{ y}^{-1}$, respectively). Nitrous oxide
29 emissions from anaerobic lagoons ($0.9 \pm 0.5 \text{ kg N}_2\text{O hd}^{-1} \text{ y}^{-1}$) and barns ($10 \pm 6 \text{ kg N}_2\text{O hd}^{-1} \text{ y}^{-1}$)
30 were unexpectedly large. Modeled methane emissions underestimated field-measurement means
31 for most manure management practices. Modeled nitrous oxide emissions underestimated field-
32 measurement means for anaerobic lagoons and manure piles, but overestimated emissions from
33 slurry storage. Revised emissions factors nearly doubled slurry CH_4 emissions for Europe and
34 increased N_2O emissions from solid piles and lagoons in the US by an order of magnitude. Our
35 results suggest that current greenhouse gas emission factors generally underestimate emissions
36 from dairy manure and highlight liquid manure systems as promising target areas for greenhouse
37 gas mitigation.

38

39 **Introduction**

40 Animal agricultural currently accounts for 20% of non- CO_2 greenhouse gas (GHG) emissions
41 globally (EPA, 2012). The majority of these emissions are derived from enteric fermentation by

42 ruminants, especially beef and dairy cattle; however, as livestock agriculture is industrialized,
43 manure management contributes an increasingly large proportion of GHG emissions. This is
44 particularly the case for dairy production which, unlike beef production, occurs predominantly
45 on feedlots in most industrialized countries. In the US, approximately 43% of CH₄ emissions
46 from dairies were from manure management (USDA, 2011), whereas in California, the state with
47 the greatest dairy production, 54% of dairy CH₄ was estimated to come from manure
48 management (CARB, 2011a). Manure management can also be an important source of nitrous
49 oxide (N₂O) emissions, accounting for an estimated 5% of global (EPA, 2012) and US (EPA,
50 2013a) N₂O emissions. Modeling estimates suggested that N₂O emissions from manure
51 management globally played a dominant role in the atmospheric increase in N₂O over the last
52 140 y (Davidson, 2009).

53
54 Emissions from dairy manure management are challenging to measure and model due to the
55 variability in management systems. Greenhouse gas sources associated with manure
56 management include solid and liquid manure storage systems and dairy surfaces in corrals and
57 barns (Figure 1). To facilitate estimates of GHG emissions from dairies, the Intergovernmental
58 Panel on Climate Change (IPCC) developed a Tier 2 model based on emission factors based on
59 manure composition, manure production rates, biogeochemical reaction rates, temperature, pH,
60 and moisture content (IPCC, 2006). Emission factors developed by the IPCC for dairies were
61 largely based on a few lab or pilot-scale studies (IPCC, 2006; Sedorovich *et al.*, 2007; Chadwick
62 *et al.*, 2011). However, the relationship between small-scale studies and actual field emissions is
63 poorly constrained, with only one study making a qualitative comparison (Jungbluth *et al.*,

64 2001). Comparing emission rates calculated using the Tier 2 model with field measurements
65 provides a valuable test of current emission factors.

66
67 Previous reviews of animal agriculture emissions have pooled a variety of livestock systems and
68 scales of studies, i.e. laboratory, pilot, and field scales (Jungbluth *et al.*, 2001; Monteny *et al.*,
69 2001; Sedorovich *et al.*, 2007; Chadwick *et al.*, 2011; Borhan *et al.*, 2012). However, GHG
70 emissions from dairies likely differ from other livestock industries due to differences in animal
71 and manure management. For example, California dairy feedlots had 1.84 million milk cows
72 and 0.78 million dairy heifers in 2009 which produced 16.4 million kg of volatile solids d^{-1} and
73 0.92 million kg N d^{-1} , over 55% of which was managed in anaerobic lagoons (CARB, 2011b). In
74 contrast, California beef feedlots had 0.46 million heifers and steers which produced just 0.85
75 million kg volatile solids d^{-1} and 0.07 million kg N d^{-1} , 1% of which was managed as liquid
76 slurry (CARB, 2011b). Accurately estimating the GHG production from this large stock of dairy
77 manure is critical for designing successful climate change mitigation programs.

78
79 The goals of this study were to synthesize a global dataset on GHG emissions from manure
80 management on dairies, compare the data with modeled values, and identify the greatest
81 mitigation opportunities. Published field measurements of CH₄ and N₂O emissions from on-
82 dairy manure management on working and research dairies globally were reviewed. Carbon
83 dioxide emission rates were also compiled but because they are not considered to contribute to
84 climate change (IPCC, 2006; with some contention, e.g., Goodland, 2013) they are not discussed
85 further. We compared mean emission rates from the field data with values calculated using the
86 IPCC Tier 2 model to identify discrepancies between measured and modeled values. We used

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

89

90 **Sources of GHG on dairies**

91 Many areas on dairies are potential sources of GHGs, in addition to the direct emissions from
92 cows (Figure 1). Manure is stored in solid or liquid form. Solid manure piles are composed of
93 the solids scraped from dairy surfaces (manure and bedding) and/or the solids separated from
94 slurry. They are heterogeneous in composition and can have both aerobic and anaerobic zones
95 within the piles, depending on moisture content and management practices. Liquid manure
96 systems were split into two groups: 1) anaerobic lagoons and 2) slurry tanks and settling ponds,
97 following the approach of the IPCC (2006). Lagoons are earthen and hold the liquid fraction
98 after mechanical or gravity-driven separation of the manure plus wash water. They are not
99 stirred and anaerobic conditions develop rapidly. Slurry tanks and settling ponds are filled with
100 unseparated, minimally-diluted manure. Slurry manure has more solids than anaerobic lagoon
101 contents, some of which typically floats on the surface and forms a surface crust. The crust is
102 important because it provides a substrate that spans anaerobic and aerobic environments where
103 N₂O production and CH₄ oxidation can both occur (Petersen *et al.*, 2005; Petersen & Sommer,
104 2011). Anaerobic digesters are another liquid manure management system, but no studies have
105 attempted to measure greenhouse gas emissions from functioning anaerobic digesters. Biogas
106 production from anaerobic digesters has been widely studied, but the literature has focused on
107 the potential reduction compared to other manure storage, rather than quantifying greenhouse gas
108 emissions from the anaerobic digester systems themselves. Emissions from digesters are likely
109 to be dependent upon the type of system and operation practices (ex. retention times,

110 effectiveness of seals in preventing gas escape, and composition of material entering the
111 digester) (Massé *et al.*, 2011; Tauseef *et al.*, 2013).

112
113 Corrals included dry lots, loafing pens, and hardstandings. Dry lot corrals are dirt-floored pens
114 in which manure is deposited and occasionally scraped into piles and/or removed. Loafing pens
115 are commonly dirt-floored and spread with some sort of bedding material, often dried manure
116 solids. Milk cows are in loafing pens only when they are not in the milking parlor, dry lots, or
117 freestalls, thus, loafing pens do not accumulate much manure. Some pasture-based dairies use
118 standoff pads to hold cows during wet periods when the cows can't be on the pastures. These are
119 small corrals in which a thick (60-100 cm) layer of sawdust and bark chips is laid over plastic
120 sheeting (Luo & Saggar, 2008). The sheeting allows the leachate from the pad to be collected
121 and treated in liquid storage systems. Hardstandings are areas with solid surfaces, such as
122 concrete, which may be used as corrals or as temporary holding pens, depending on their size
123 and location on the dairy.

124
125 Barns were measured as entire barns or only barn floors, depending on measurement approach.
126 Measurements of whole barns include pens and/or freestalls, manure removal and feeding alleys,
127 and often the cows themselves. Barn floors are heterogeneous and typically have paved or
128 slatted-floor areas for livestock movement, farmer access, and manure management, as well as
129 stalls or pens with some sort of soft bedding where the cows can rest. Emissions for entire
130 dairies were reported by two studies and were also included.

131

132 **Field measurement data compilation**

133 Thirty-eight studies met our criteria (Table 1), most of which were located in North America and
134 Europe (Figure 2, Table S1). Emission rates were measured using flux chambers or
135 micrometeorological techniques. Measurement techniques varied by dairy source area, which
136 was expected given the different spatial scales (piles versus whole barns) or materials (liquid
137 versus solid) involved. Measurements were typically carried out every 1 to 2 months over 1 to 5
138 days for up to a year. Data compiled from the studies included farm characteristics such as the
139 surface area of the pens and lagoons, and number of cows, as available; measurement and gas
140 analysis technique; sampling duration and frequency; and climate data as mean annual
141 temperature [MAT], mean annual precipitation [MAP], and temperature during sampling (Tables
142 S2-S8). A difficulty in comparing literature data was the difference in, or lack of, information
143 reported. When possible we remedied this by contacting the authors or providing reasonable
144 estimates of missing information. Missing MAP and MAT data were estimated using data from
145 the nearest city on www.worldclimate.com. Air temperature during sampling periods (“sampling
146 temperature”) was estimated using either the monthly averages from www.worldclimate.com or
147 the almanac feature on www.wunderground.com. Methane fluxes that included enteric
148 fermentation-derived emissions from barns, corrals, or whole dairies were corrected for enteric
149 emissions by subtracting the IPCC regional estimate for enteric fermentation (IPCC, 2006).
150 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
151 European and modern, high-producing Chinese dairies (rather than the default of $68 \text{ kg hd}^{-1} \text{ d}^{-1}$
152 for Asia, which assumes low-producing cows on small farms).

153
154 Most studies included measurements of the same area at different times (e.g., seasonally) and/or
155 measurements from areas in which some management component was different (e.g., mixed vs.

156 static manure piles, barns with different flooring and scraping mechanisms). Each permutation
157 was included in the compilation. The mean emission rate for a given dairy area was calculated
158 by first averaging the emission rates compiled from each paper, then averaging those values,
159 such that n is the number of studies rather than the number of measurements. This method
160 avoided weighting the mean towards studies, management practices, and measurement
161 techniques with more measurements. Some studies used climate data to extrapolate between
162 measurements to calculate an annual emission rate. We included these annual estimates in the
163 appendices, but they were excluded from the calculation of mean emission rates and the
164 statistical analyses. Statistical analyses were performed using JMP 10.0.2 (SAS Institute, 2012).
165 Correlations between GHG emission rates, climate variables, cow populations, manure volume,
166 and other variables in Tables S2-S8 were explored using multiple linear regressions, with
167 statistical significance determined as $P < 0.10$.

168
169 Measurement technique may have affected emissions measured from all manure management
170 systems; large footprint techniques generally measured higher CH₄ and N₂O emissions than
171 studies using dynamic or static chambers, with the exceptions of CH₄ from anaerobic lagoons
172 and corrals. The varied composition and oxygen availability of manure stores creates CH₄ and
173 N₂O emission hotspots in space and time which can be missed by smaller footprint techniques
174 (Parkin & Kaspar, 2004; Sommer *et al.*, 2004). Concurrent measurements using different
175 techniques have not been made (with one exception, (Sommer *et al.*, 2004)), but are needed to
176 resolve their impact on reported emissions. Because of the uncertainty in the extent of the
177 impact of measurement technique, in the analysis below we calculated mean emissions from
178 each area using all available data. In most cases, this likely produced a conservative estimate.

179

180 Emission rates are presented as the mass of trace gas emitted per head per time (kg trace gas hd⁻¹
181 y⁻¹) and per unit area per time (kg trace gas m⁻² y⁻¹). The discussion focuses on per head numbers
182 for several reasons. The goal of this study was to evaluate our ability to estimate dairy emissions
183 from manure management at regional to global scales; therefore, emissions factors needed to use
184 units that were widely known. Most countries have fairly good estimates of the number of
185 animals present, but estimates of the area of the various manure handling systems have not been
186 attempted with few exceptions (Chung *et al.*, 2013). Others (Place & Mitloehner, 2010; O'Brien
187 *et al.*, 2012) have argued that reporting emissions in terms of the mass of milk produced gives a
188 better sense of the GHG-efficiency of production. This is a useful approach for comparing
189 different production systems. However, milk production varies significantly by breed, feed, cow
190 age and stage in lactation cycle (ex. McCandlish, 1920; Zimmerman *et al.*, 1991); furthermore, it
191 is not relevant to the emissions from different manure management approaches in which we are
192 interested here. Few studies (only 11 of the 38 studies included here, with 6 of the studies
193 measuring whole barns) reported milk production.

194

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

200

201 **Summary of field measurements**

202 Anaerobic lagoons and slurry systems had the highest per head GWP on dairies, averaging 12.8
203 ± 7 Mg CO₂e hd⁻¹ y⁻¹ and 3.5 ± 1.7 Mg CO₂e hd⁻¹ y⁻¹, respectively (Table 2a). Mean lagoon
204 GWP was about 20 times higher than mean solid manure storage GWP. When expressed on an
205 area basis, lagoons and slurry systems were similar, averaging 703 ± 195 kg CO₂e m⁻² y⁻¹ and
206 827 ± 320 kg CO₂e m⁻² y⁻¹, respectively (Table 2b). These rates were high; for comparison, the
207 highest landfill CH₄ emissions rates reported in Bogner et al. (1995) were 248 kg CO₂e m⁻² y⁻¹,
208 less than half those from liquid manure systems. Barn floors had the lowest GWP (38 ± 7 kg
209 CO₂e hd⁻¹ y⁻¹) of all the dairy environments studied. Methane emissions were the largest
210 component of total GWP for all sources except for barns and corrals.

211

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

217

218 Barns had the greatest N₂O emissions by nearly an order of magnitude, with 10.3 ± 6.2 kg N₂O
219 hd⁻¹ y⁻¹ (Table 2a), although field data were highly variable (Table S6). Corrals and solid
220 manure piles were the next largest N₂O source with 1.5 ± 0.8 kg N₂O hd⁻¹ y⁻¹ and 1.1 ± 0.7 kg
221 N₂O hd⁻¹ y⁻¹, respectively (Table 2a). Nitrous oxide emissions from anaerobic lagoons and slurry
222 stores were also substantial, with 0.9 ± 0.5 kg N₂O hd⁻¹ y⁻¹ and 0.3 ± 0.3 kg N₂O hd⁻¹ y⁻¹,
223 respectively. The relatively large net N₂O flux from liquid manure storage was surprising given
224 the predominantly anaerobic conditions typical of unaerated systems. Nitrogen in liquid manure

225 is mostly in the form of ammonium (NH_4^+) and organic N (Harter *et al.*, 2002), and though
226 anaerobic lagoons are generally anaerobic, aerobic conditions which could promote
227 denitrification exist at inlets. Other N_2O formation reactions are also feasible, such as
228 denitrification of nitrate (NO_3^-) produced through annamox (anaerobic NH_4^+ oxidation, (Mulder
229 *et al.*, 1995; Maeda *et al.*, 2010)), Feammox (anaerobic NH_4^+ oxidation coupled to Fe reduction,
230 (Yang *et al.*, 2012)), or Mnammox (anaerobic NH_4^+ oxidation coupled to Mn reduction,
231 (Engström *et al.*, 2005)). Hardstandings and barn floors, surfaces which were scraped or flushed
232 frequently, had CH_4 and N_2O emissions generally one to three orders of magnitude lower than
233 the other sources. These trends were consistent between the per-head and per-area data (Table
234 2b) and showed that the type of storage or surface measured was the greatest factor controlling
235 emission rates.

236
237 Methane emissions from soils are known to be temperature dependent (Conrad, 2007) and
238 models often assume that manure CH_4 emissions are positively correlated with MAT (Mangino
239 *et al.*, 2002; IPCC, 2006). Individual field studies observed greater CH_4 emissions in summer
240 and/or with warmer sampling temperatures for manure piles, barns, and whole dairies; however,
241 there was no significant correlation between CH_4 emissions and temperature when all the studies
242 for a given source area were considered. The lack of correlation for liquid systems may be due
243 to the limited range of MAT represented by the field studies; all studies but one (Todd *et al.*,
244 2008) sampled liquid systems that were in regions where MAT was 6–15°C. Air temperature
245 during sampling had a larger range (-10.6 to 34.4°C, Tables S2-S3), but overall liquid systems in
246 warm climates were under-represented. Differences in volatile solid content, the other key factor
247 determining CH_4 production (Mangino *et al.*, 2002), may have also confounded any temperature

248 effect. Insufficient data were available to test the effect of volatile solid content on CH₄
249 emissions within or across studies.
250
251 Methane and N₂O emissions were strongly correlated to each other for solid manure piles ($r^2 =$
252 $0.73, P < 0.001$) and weakly correlated for corrals ($r^2 = 0.26, P < 0.08$). This suggests that in
253 solid manure management systems, at least a portion of the N₂O fluxes were derived from
254 denitrification, which requires the same general environmental conditions as methanogenesis
255 (warm temperatures, abundant labile C, anaerobic conditions). Corral CH₄ emissions were
256 negative, indicating soil uptake, in 5 out of 18 cases (Table S4); negative fluxes occurred in late
257 summer when the soils were dry, or in winter when the soils were cold or frozen. The highest
258 corral N₂O emissions were measured in late spring when a combination of warmer temperatures
259 and moist soils likely promoted nitrification and denitrification (Table S4). The lowest values
260 were < 50% of the highest emissions and occurred in late fall and winter (Table S4). Despite
261 these seasonal patterns, neither CH₄ nor N₂O emissions from corrals were correlated with
262 temperature and/or precipitation. Leytem et al. (2011) measured higher N₂O emission rates from
263 manure piles in warmer months (May and June) than colder ones (September and March, Table
264 S5), but no correlations were found when all manure pile data were pooled.
265
266 Specific management practices could have made it difficult to detect a temperature effect if one
267 existed. Mixing solid manure piles resulted in increased CH₄ and N₂O emissions (Yamulki,
268 2006; Maeda *et al.*, 2010; Ahn *et al.*, 2011; Leytem *et al.*, 2011), contrary to expectations that
269 mixing would aerate the pile and decrease CH₄ production. The addition or accumulation of
270 fresh manure was another source of emissions. Addition of fresh material increased pile

271 emissions (Leytem *et al.*, 2011) and the accumulation of fresh material in corrals was likely one
272 of the most important factors driving positive CH₄ fluxes. Borhan et al. (2011a) measured
273 greater CH₄ and N₂O emissions from a dry lot corral than from loafing pens (Table S4), probably
274 due to the corrals having a greater influx of fresh manure and localized, high-moisture urine
275 patches. Methane emissions from the brick hardstanding were relatively high in the summer
276 (Table S7) (Gao *et al.*, 2011), likely because scraping was less frequent compared to other dairies
277 (every 1–4 weeks vs. daily for most other hardstandings, Table S7). Accordingly, Adviento-
278 Borbe et al. (2010) observed a significant, positive correlation between CH₄ emissions and
279 manure depth on the barn floor. However, Gao et al. (2011) was the only study of hardstandings
280 to use an open path laser rather than flux chambers.

281

282 **Emission rate modeling**

283 The field measurements provide a test of emission rate models. The IPCC Tier 2 approach
284 models CH₄ emissions (EF_{CH_4} , g CH₄ hd⁻¹ y⁻¹) based on the volatile solid production by the cows
285 (VS, kg VS hd⁻¹ y⁻¹), a CH₄ conversion factor (MCF , %) for the manure management practice,
286 and the maximum possible CH₄ production rate from the volatile solids in the manure (B_o , m³
287 CH₄ kg VS⁻¹):

288

$$289 \quad EF_{CH_4} = VS \times MCF / 100 \times B_o \times 662 \text{ g CH}_4 \text{ m}^{-3} \text{CH}_4 \quad (1).$$

290

291 Volatile solid production by cows can be determined from manure analysis (where volatile solids
292 are the combustible components of solid manure) or estimated based on feed intake rate,
293 digestibility, and dry matter content. No studies reviewed here included all the information

294 necessary to calculate dairy specific *VS* so we used the IPCC regional values for international
295 data (IPCC, 2006) and averages of state values for the US (EPA, 2013b).

296

297 The IPCC Tier 2 approach models direct N₂O emissions based on annual N excretion rates,
298 which themselves are a function of energy intake by the cows, crude protein content of feed,
299 milk production rate, milk protein content, cow growth, typical animal mass, and an emission
300 factor (EF_{N_2O} , kg N₂O-N kg N excreted⁻¹) (equations 10.31, 10.32, and 10.33 in IPCC, 2006).

301 EF_{N_2O} can be converted into N₂O emission rates equivalent to those measured here (N_2O_D , g N₂O
302 hd⁻¹ d⁻¹, where the subscript *D* refers to direct emissions) using the typical animal mass (*TAM*,
303 kg) and country- or region-specific N excretion rates (N_{ex} , kg N 1000 kg *TAM*⁻¹ d⁻¹):

304

$$305 \quad N_2O_D = EF_{N_2O} \times 44/28 \times TAM \times N_{ex} \quad (2).$$

306

307 In our calculations, we used a *TAM* of 600 kg, the default for Western Europe (but similar to the
308 North American default value of 604 kg) (IPCC, 2006). Indirect N₂O emissions, derived from
309 the oxidation of gaseous emissions such as ammonia (NH₃) and nitrous oxides (NO_x), are
310 important for calculating the amount of N remaining in manure for its use as an organic fertilizer
311 (IPCC, 2006). We did not include indirect N₂O emissions in our calculations, and thus what is
312 reported here should be considered minimum estimates.

313

314 We used the field measurement means and equations 1 and 2 to derive revised *MCF*s and EF_{N_2O} s
315 for the source areas. This is the first time broadly applicable, field measurement-derived *MCF*s

316 and $EF_{N_2O_S}$ have been calculated. Some revised MCF s and $EF_{N_2O_S}$ were very different from
317 current values.

318

319 **Comparisons with modeled emissions**

320 *Measured vs. modeled CH₄ emissions*

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

330

331 The impact of the revised barn/deep pit and slurry store MCF s was evaluated using data on slurry
332 storage in Europe because six of 13 barn studies were conducted in Europe, while slurry studies
333 were distributed in temperate regions globally. We used 1990 and 2011 emissions inventory
334 data for 12 European countries compiled by the United Nations Framework Convention on
335 Climate Change (UNFCCC, 2014). Three of the 15 countries in the dataset were excluded due
336 to lack of data or falling outside the cool MAT temperature zone. The European data did not
337 distinguish between slurry stored in deep pits and tanks or ponds (i.e., one MCF was used for all
338 slurry, that in deep pits and in ponds), whereas we calculated revised MCF s for each system.

339 Thus, revised European slurry CH₄ emissions were calculated using each revised *MCF* to
340 provide a range. However, deep pit storage is often a temporary holding for slurry that is
341 eventually transferred to slurry tanks or ponds, so the *MCF* for slurry stores is likely more
342 applicable.

343
344 Calculations using the revised deep pit *MCF* gave total CH₄ emissions from European slurry
345 storage that were less than those using the country-specific slurry *MCF*s (8.4 ± 4.6 Tg CO₂e y⁻¹
346 vs. 15.2 Tg CO₂e y⁻¹, respectively, Figure 4). However, the revised slurry pond *MCF* increased
347 CH₄ emissions from slurry for most countries, with total emissions of 25.9 ± 12.2 Tg CO₂e y⁻¹, a
348 gain of 10.7 Tg CO₂e y⁻¹ (Figure 4). Increases were greatest for the countries with the most
349 manure in liquid systems (Denmark, Germany, The Netherlands, Switzerland, and Sweden). We
350 found a similar trend using detailed data for the Netherlands (RIVM *et al.*, 2013), with modeled
351 slurry CH₄ emissions two times larger than those estimated in the current inventory (data and
352 calculations not shown).

353
354 The uncertainty in slurry *MCF* has consequences for the evaluation of the European dairy
355 industry's progress in mitigating its GHG emissions. Between 1990 and 2011, the 13 countries
356 considered here decreased the total number of cows by nearly 8 million hd leading to a
357 corresponding decrease in emissions from enteric fermentation by 515 Gg CH₄ (Table 5). The
358 reduction in cows also decreased VS production by 7.8 Tg so there was less manure to manage
359 and produce GHG. However, an increase in the proportion of manure in liquid management in
360 most countries offset some of this decrease in CH₄ production; the current estimates suggest a
361 total net decrease (combined change in enteric and manure management emissions) of 480 Gg

362 CH₄ (Table 5). Using the revised slurry *MCF* for the 2011 estimates gives a smaller total net
363 decrease of 166 Gg CH₄, with some countries (Denmark, Switzerland, and The Netherlands)
364 having net *increases* of 30–50 Gg CH₄ rather than decreases (Table 5).

365

366 *Measured vs. modeled N₂O emissions*

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

373

374 The impact of revised *EF_{N2O}* values was evaluated using state-specific data from 2011 for the US
375 (Table 7) because eight of the nine anaerobic lagoon studies and five of the ten manure pile
376 studies occurred in the US or North America; therefore, the revised *EF_{N2O}* values should be
377 applicable to this region. The EPA assumed zero N₂O emissions from anaerobic lagoons,
378 whereas the revised *EF_{N2O}* gave 1.79 ± 0.90 Tg CO₂e y⁻¹ (Figure 5). Nitrous oxide emissions
379 from solid manure piles also increased from 0.51 Tg CO₂e y⁻¹ to 3.36 ± 2.04 Tg CO₂e y⁻¹ using
380 the revised *EF_{N2O}* (Figure 5). Combined, the revised values increased manure management N₂O
381 emissions in the US by more than 4.5 Tg CO₂e y⁻¹, 25% of the 2011 estimate of 17.3 Tg CO₂e
382 (EPA, 2013a).

383

384 Whole barn N₂O emissions varied widely between studies, and the measurements of Leytem et
385 al. (2013) and Samer et al. (2012) suggested an order of magnitude increase in EF_{N_2O} . They also
386 indicated that barns may be significant, largely unaccounted sources of N₂O from dairies (2 to 3
387 times more kg N₂O hd⁻¹ y⁻¹ than corrals or solid piles). No standard model has been established
388 for calculating N₂O emissions from barns that do not have deep pit manure storage. If we
389 assume that two-thirds of the cows in the 13 European countries in Table 4 were kept in barns
390 (11 million hd) for half of the year, and emitted 1 kg N₂O hd⁻¹ y⁻¹ (the approximate mean of the
391 measurements by Zhang et al. (2005), which was the lowest of the three studies that measured
392 N₂O), then barns emitted 1.64 Tg CO₂e, on the same scale as the revised N₂O emissions from
393 anaerobic lagoons in the US (Table 7). Using the field-measurement mean of 10 kg N₂O hd⁻¹ y⁻¹
394 for the calculation increased barn emissions by an order of magnitude (to 16.4 Tg CO₂e),
395 equivalent to the warming potential of slurry system-derived CH₄ in Europe (Table 4). More data
396 are needed to assess if barns are actually such large sources of N₂O.

397

398 **Discussion and conclusions**

399 Our results highlight potential issues with the application of IPCC Tier 2 models to estimate
400 GHG emissions from livestock manure. Emission factors were typically based on few studies,
401 many of which were not designed for GHG inventory estimation or were small-scale pilot or
402 laboratory experiments, and spanned various livestock systems (Jungbluth *et al.*, 2001; Chung *et al.*,
403 2013). These approaches are unlikely to accurately approximate field-scale fluxes from
404 manure management in a specific livestock system. Our review of field-based research on
405 dairies suggests that current Tier 2 model parameters are generally underestimating dairy
406 emissions.

407
408 Data were conspicuously lacking from India and China, which have the fastest growing dairy
409 industries in the world (growing by 10.7 and 7.6 million hd, respectively, between 2000 and
410 2010; FAO, 2014). Though data were not available on manure management practices in the two
411 countries, estimated CH₄ emissions for each suggest that China is treating more manure in liquid
412 form; for every million dairy milk cows gained between 2000 and 2010, India's CH₄ emissions
413 from manure management increased by 5 Gg CH₄, whereas China's increased by 9 Gg CH₄ per
414 million hd (FAO, 2014). Accurately modeling these emissions is critical for policy decisions
415 towards GHG emission reduction.

416
417 The disagreement between field measurements and modeled values provides mechanistic support
418 for discrepancies reported by airborne measurements and modeling. In a top-down approach
419 combining aircraft and tower measurements with an atmospheric transport model, Miller et al.
420 (2013) calculated total CH₄ emissions for the US that were 1.5 times greater than the EPA
421 bottom-up approach. Underestimation of emissions from fossil fuel extraction was responsible
422 for a significant part of this discrepancy, but emissions from livestock enteric fermentation and
423 manure management were calculated to be twice that of the EPA estimate (Miller *et al.*, 2013).
424 A smaller scale analysis for the Los Angeles Basin found similar dairy CH₄ fluxes between top-
425 down and bottom-up approaches (Peischl et al., 2013).

426
427 Despite the uncertainties in emissions inventories described above, targets for GHG reduction
428 can be identified. As shown by the European example, decreasing the number of cows can
429 reduce GHG emissions by decreasing both enteric fermentation and manure production (Ripple

430 *et al.*, 2013). While this is the trend in developed countries, developing nations have growing
431 livestock populations which must be managed appropriately to be sustainable (Eisler *et al.*,
432 2014). The most effective GHG mitigation approach for manure management depends on how
433 manure is handled and stored. Where liquid manure management systems are common,
434 particularly anaerobic lagoons which were the highest total and per-cow CH₄ source, they
435 represent the greatest opportunity for GHG emissions reduction. Some estimates suggest that the
436 total CH₄ and N₂O emissions per head from anaerobic digesters is about 10% of the emissions
437 from anaerobic lagoons (CARB, 2011a). The EPA (2011) estimated that adoption of anaerobic
438 digesters by all US dairies for which this technology is feasible (those with liquid manure
439 management systems and > 500 hd, or approximately 2,650 farms with 3 million hd) could
440 reduce US CH₄ emissions by 41.25 Tg CO₂e y⁻¹, or more than 85% of the total CH₄ emissions
441 from dairy manure management. In addition, these anaerobic digesters would be capable of
442 producing more than 6.8 million MWh y⁻¹ (EPA, 2011). According to the GHG equivalency
443 calculator at www.epa.gov, this would offset an additional 17.6 Tg CO₂e of CO₂ emissions from
444 energy production.

445

446 Our results show significant disagreement between measured and modeled GHG emissions from
447 dairies globally. Revised emission factors based on the field data led to greater estimated GHG
448 emissions from the US and Europe. More field data are needed to refine these models. To
449 maximize the usefulness of field measurements, better reporting of herd characteristics (number
450 of milk cows and heifers, average mass, milk production, dry matter and N intake), dairy
451 characteristics (manure handling practices and storage dimensions, climate parameters, available
452 land for manure spreading, typical management schedule), and manure characteristics (amount

453 handled by each storage method, volume, volatile solid content, C and N content, manure
454 temperature) are needed. Emission rates must be reported with the data necessary to convert
455 between units of per head, per area, and per kg milk production in addition to units specific to
456 certain types of sources, such as per HPU for barns or per volume for liquid storage. Future
457 research should focus on GHG emissions from several major dairy industries, particularly China
458 (the fastest-growing), India (the largest), and California (the largest in the US) (FAO *et al.*,
459 2006). Each region has unique issues related to climate, development, and legislation that
460 complicate estimating GHG emissions without direct measurements. Furthermore, longer
461 monitoring periods are needed to disentangle the effects of management and climate on
462 emissions and enable more accurate estimates of annual averages.

463

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469

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- 692

693 **Supporting Information**

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

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

696 Table S2. Anaerobic lagoon study characteristics and GHG emissions.

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

698 Table S4. Corral study characteristics and GHG emissions.

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

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

701 Table S7. Hardstanding study characteristics and GHG emissions.

702 Table S8. Whole dairy study characteristics and GHG emissions.

703

704 **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)	x			x	x		x	
Borhan et al. (2011b)	x				x		x	
Bjorneberg et al. (2009)	x				x			
Leytem et al. (2011)	x			x	x			
Leytem et al. (2013)	x						x	
Craggs et al. (2008)	x							
Safley and Westerman (1988)	x							
Safley and Westerman (1992)	x							
Todd et al. (2011)	x							
Husted (1994)		x	x					
Sneath et al. (2006) ^a		x	x					
Hensen et al. (2006)		x						x
Kaharabata et al. (1998)		x						
Kahn et al. (1997)		x						
VanderZaag et al. (2011)		x						
Ahn et al. (2011)			x					
Amon et al. (2006)			x					
Brown et al. (2002)			x					
Gupta et al. (2007)			x					
Osada et al. (2001)			x					
Sommer et al. (2004)			x					
Kaharabata et al. (2000)					x			
Luo and Sagar (2008)					x			
Ellis et al. (2001)						x		
Gao et al. (2011)						x		
Misselbrook et al. (2001)						x		
Adviento-Borbe et al. (2010)							x	
Kinsman et al. (1995)							x	
Marik and Levin (1996)							x	
Ngwabie et al. (2009)							x	
Ngwabie et al. (2011)							x	
Samer et al. (2012)							x	
Snell et al. (2003)							x	
van Vliet et al. (2004)							x	
Wu et al. (2012)							x	
Zhang et al. (2005)							x	
Zhu et al. (2012)							x	
McGinn and Beauchemin (2012)								x

^a excludes slurry tank work which was pilot-scale

705
706

707 **Table 2a.** Summary of the means and ranges of N₂O and CH₄ emission rates measured by the studies listed in Table 1, in kg hd⁻¹ y⁻¹.

	Emission rate (kg hd ⁻¹ y ⁻¹)			Emission global warming potential ^b (kg CO ₂ e hd ⁻¹ y ⁻¹)		
	mean ^a ± standard error (n)			mean ^a ± standard error (n)		
	CH ₄	N ₂ O	CO ₂	from CH ₄	from N ₂ O	total (CH ₄ +N ₂ O)
Anaerobic lagoons	368 ± 193 (9) 4 - 2814	0.9 ± 0.5 (4) 0.004 - 3.9	687 ± 266 (6) 4.8 - 2400	12510 ± 7334	264 ± 131	12775 ± 6699
Slurry stores	101 ± 47 (6) 0 - 328	0.3 ± 0.3 (3) 0 - 4.5	nm	3422 ± 1601	81 ± 76	3504 ± 1680
Solid^c	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^d	-17 ± 24 (6) -128 - 210	1.5 ± 0.8 (4) 0.0 - 12	4242 ± 3040 (3) 134 - 20292	-577 ± 844	454 ± 272	-124 ± 1073
Hard-standings^d	1.2 ± 0.8 (3) -3.8 - 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^d	33 ± 19 (10) -61 - 289	10 ± 6 (3) 0 - 22	7204 ± 5507 (3) 273 - 35058	1120 ± 931	3076 ± 3154	4197 ± 2496
Whole dairy^d	96 ± 35 (18) -91 - 350	nm	nm	3252 ± 1191	nm	3252 ± 1194

^a 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).

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

^c Excludes data from Gupta et al. (2007)

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

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710 **Table 2b.** Summary of the means and ranges of N₂O and CH₄ emission rates measured by the studies listed in Table 1, in kg m⁻² y⁻¹,
 711 with same footnotes as Table 2a.

	Emission rate (kg m⁻² y⁻¹)			Emission global warming potential^b (kg CO₂e m⁻² y⁻¹)
		mean ^a ± standard error (n) range		mean ^a ± standard error (n)
	CH₄	N₂O	CO₂	total (CH₄+N₂O)
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^c	2.8 ± 0.9 (6) 0 - 13	0.3 ± 0.1 (7) 0.005 - 1.0	268 ± 103 (4) 13 - 461	147 ± 49
Corrals^d	-0.8 ± 0.9 (6) -5.2 - 3.8	0.03 ± 0.014 (4) 0.001 - 0.22	83 ± 52 (3) 12 - 365	-16 ± 35
Hard-standings^d	0.5 ± 0.6 (3) -0.11 - 2.2	0.0003 ± 0.0001 (2) 0 - 0.001	nm	18 ± 21
Barn floor	0.2 ± 0.2 (4) 0 - 1.0	0.01 ± 0.01 (4) 0.00 - 0.04	21 ± 10 (4) 5 - 58	9.4 ± 7.0
Whole barn^d	3.4 ± 2.0 (10) -2.8 - 31	0.9 ± 0.7 (3) 0 - 2.3	774 ± 578 (3) 39 - 3713	381 ± 277
Whole dairy^d	nm	nm	nm	nm

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

	<i>VS</i>		<i>MCF</i>		<i>B_o</i>		<i>CH₄ emission rate</i>		
	(<i>kg-VS</i> <i>hd⁻¹ y⁻¹</i>)	<i>VS</i> data source	(%)	<i>MCF</i> data source	(<i>m³ CH₄</i> <i>kg VS⁻¹</i>)	<i>B_o</i> data source	Modeled (<i>kg hd⁻¹ y⁻¹</i>)	Field (<i>kg hd⁻¹ y⁻¹</i>)	Field- derived MCF (%)
Anaerobic lagoon	2770	average ID, NM, TX, NC ^a	74	average ID, NM, TX, NC ^a	0.24	default ^{ab}	326	368 ± 193 (9)	84 ± 44
Slurry stores	1861.5	Western Europe ^b	11	MAT=11, with crust ^b	0.24	Western Europe ^b	33	101 ± 47 (6)	34 ± 16
Manure pile	2750	average TX and ID ^a	5	static pile ^a	0.24	default ^a	22	13 ± 11 (4)	2.9 ± 2.5
Corrals	2800	average TX and ID ^a	1	cool MAT ^{ab}	0.24	default ^{ab}	4.4	-17 ± 24 (6)	-3.8 ± 5.5
Paved surfaces and barn floors	1861.5	Western Europe ^b	1	cool MAT ^b	0.24	Western Europe ^b	3.0	0.9 ± 0.5 (6)	0.31 ± 0.16
Brick hardstanding	1022	Asia ^b	1	cool MAT ^b	0.13	Asia ^b	0.9	1.6 ± 7.7 (1)	1.8 ± 8.8
Whole barn	1861.5	Western Europe ^b	3	deep pit, cool MAT ^b	0.24	Western Europe ^b	8.9	33 ± 19 (10)	11 ± 6.4
Whole dairy (= corral + manure pile + slurry tank + whole barn)							56	96 ± 35 (18)	

716 ^a from (EPA, 2013b)

717 ^b from (IPCC, 2006)

718

719 **Table 4.** Comparison of modeled slurry emissions in 13 cool MAT European countries using the 2011 liquid slurry *MCF* s (and other
720 inputs) for each country (UNFCCC, 2014), the revised deep pit and slurry *MCF*s from Table 3. Countries with 60% liquid manure
721 management are Denmark, Germany, The Netherlands, Switzerland and Sweden.

	Dairy cows (1000 hd)	VS (kg DM/hd/d)	Bo (m3 CH4/kg VS)	Liquid fraction (%)	Liquid <i>MCF</i> (%)	Revised <i>MCF</i> (deep pit) (%)	Revised <i>MCF</i> (slurry) (%)	Liquid manure emissions (Gg CH ₄ y ⁻¹)			
								Using current <i>MCF</i>	Using revised deep pit <i>MCF</i>	Using revised slurry <i>MCF</i>	
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 (Tg CO ₂ e)	6.6	5.4 ± 2.9	16.7 ± 7.8

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724 **Table 5.** Comparisons between 1990 and 2011 data for the 13 countries in Table 4 (UNFCCC, 2014). Negative values indicate a
 725 decrease from 1990 to 2011. The revised slurry *MCF* was used to recalculate 2011 emissions, not the revised deep pit *MCF*, and the
 726 1990 value were not adjusted.

	Cows (1000 hd)	VS production (Gg)	Liquid fraction (%)	Enteric fermentation emissions (Gg CH ₄)	Manure management emissions (default <i>MCF</i>) (Gg CH ₄)	Net emissions (default <i>MCF</i>) (Gg CH ₄)	Manure management emissions (revised <i>MCF</i>) (Gg CH ₄)	Net emissions (revised <i>MCF</i>) (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 CO₂e	-17.51	1.16	-16.35	11.87	-5.64

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728 **Table 6.** Nitrous oxide emissions modeling inputs and results. EF_{N_2O} = emissions factor, TAM = typical animal mass, N_{ex} = country-
 729 or region-specific N excretion rates. EF_{N_2O} uncertainty range is a factor of 2 for all but anaerobic lagoons and whole barns.

	EF_{N_2O} (kg N ₂ O-N kg N excreted ⁻¹)	TAM (kg hd ⁻¹)	N_{ex} (kg N 1000 kg TAM ⁻¹ d ⁻¹)	EF_{N_2O} and N_{ex} sources	N ₂ O emission rate		Field-derived EF_{N_2O} (kg N ₂ O-N kg N excreted ⁻¹)
					Modeled (kg hd ⁻¹ y ⁻¹)	Field (kg hd ⁻¹ y ⁻¹)	
Anaerobic lagoon	0	600	0.25	average of ID and TX ^a	0	0.9 ± 0.5 (4)	0.010 ± 0.005
Slurry tanks and ponds	0.005	600	0.44	North America, with crust ^b	0.8	0.3 ± 0.3 (3)	0.005 ± 0.005
Manure pile	0.005	600	0.26	ID ^a	0.4	1.1 ± 0.7 (4)	0.033 ± 0.020
Corrals	0.02	600	0.25	average of ID and TX ^a	1.7	1.5 ± 0.8 (4)	0.048 ± 0.026
Barn floors and paved surfaces	0.02	600	0.44	North America ^b	3.0	0.02 ± 0.01 (6)	0.0001 ± 0.0001
Whole barn (deep pit)	0.002	600	0.48	western Europe ^b	0.3	10 ± 6 (3)	0.062 ± 0.038

730 ^a from (EPA, 2013b)

731 ^b from (IPCC, 2006)

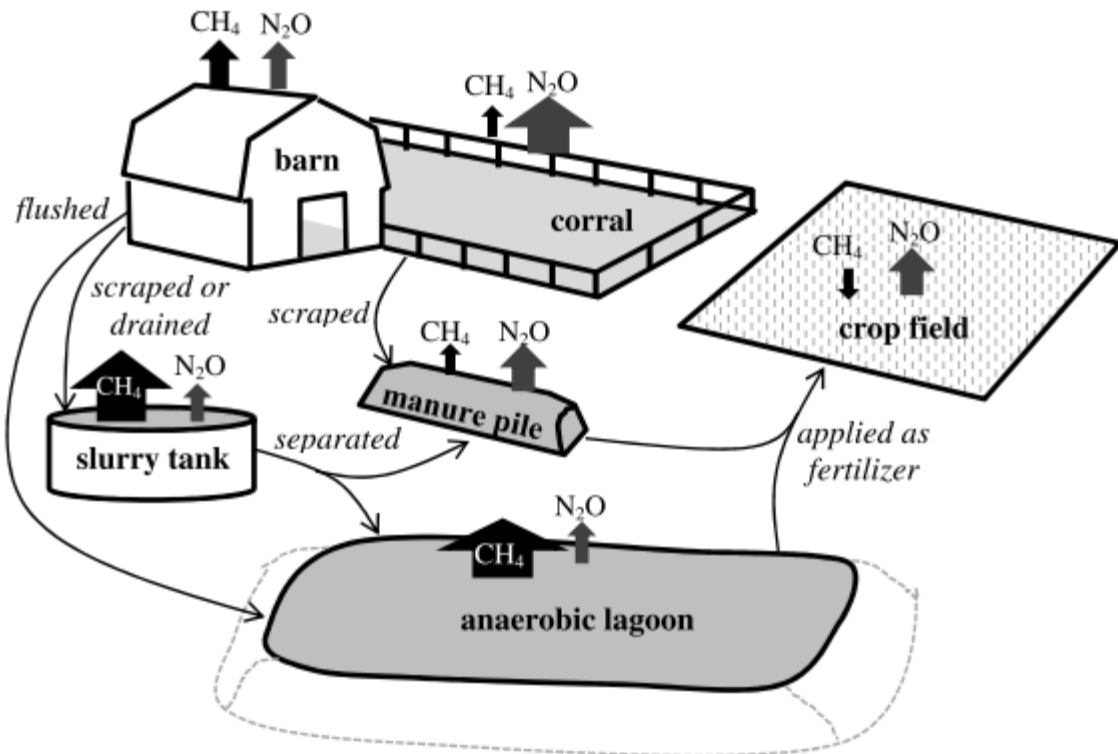
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733 **Table 7.** Comparison of N₂O emissions from anaerobic lagoons and solid manure piles modeled for the US using the current EF_{N_2O} (0
734 and 0.005 kg N₂O-N kg N excreted⁻¹, respectively) (EPA, 2013a) and the revised EF_{N_2O} (0.01±0.005 and 0.033 ± 0.02 kg N₂O-N kg
735 N excreted⁻¹, respectively) from Table 5. For all states, dairy cow $TAM = 680$ kg, heifer $TAM = 407$, and heifer $N_{ex} = 69$ kg N 1000
736 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 y ⁻¹)	EPA solid pile N ₂ O emissions (kg N ₂ O y ⁻¹)	revised solid pile N ₂ O emissions (kg N ₂ O y ⁻¹)	EPA	
								lagoons N ₂ O emissions (kg N ₂ O y ⁻¹)	revised lagoons N ₂ O emissions (kg N ₂ O y ⁻¹)
Alabama	11000	6000	7	16	130	627	4141 ± 2510	0	2869 ± 1434
Alaska	500	200	34	24	128	131	866 ± 525	0	185 ± 93
Arizona	185000	60000	9	61	159	15336	101217 ± 61344	0	207887 ± 103944
Arkansas	12000	4000	10	9	122	870	5745 ± 3482	0	1567 ± 783
California	1750000	760000	9	59	158	148050	977127 ± 592198	0	1941095 ± 970547
Colorado	123000	75000	11	64	159	13314	87874 ± 53257	0	154930 ± 77465
Connecticut	19000	9500	16	13	145	2691	17757 ± 10762	0	4372 ± 2186
Delaware	5000	3000	19	10	143	852	5621 ± 3406	0	896 ± 448
Florida	114000	30000	7	43	149	6816	44986 ± 27265	0	83741 ± 41871
Georgia	78000	23000	9	23	147	5970	39404 ± 23881	0	30515 ± 15257
Hawaii	2000	1000	9	57	130	145	956 ± 580	0	1835 ± 918
Idaho	574000	320000	11	65	157	60731	400822 ± 242922	0	717725 ± 358863
Illinois	98000	46000	39	16	146	33772	222897 ± 135089	0	27711 ± 13855
Indiana	172000	62000	29	24	150	43943	290021 ± 175770	0	72733 ± 36366
Iowa	210000	130000	34	20	152	67738	447069 ± 270951	0	79692 ± 39846
Kansas	122000	85000	21	36	151	24608	162414 ± 98432	0	84371 ± 42185
Kentucky	77000	50000	14	3	134	9262	61132 ± 37050	0	3970 ± 1985
Louisiana	19000	5000	10	9	125	1379	9103 ± 5517	0	2483 ± 1241
Maine	32000	17000	20	10	144	5674	37450 ± 22697	0	5674 ± 2837
Maryland	53000	28000	22	8	144	10330	68179 ± 41320	0	7513 ± 3756
Massachusetts	13500	7500	22	8	138	2554	16856 ± 10216	0	1857 ± 929
Michigan	361000	148000	24	29	158	80977	534445 ± 323906	0	195693 ± 97847

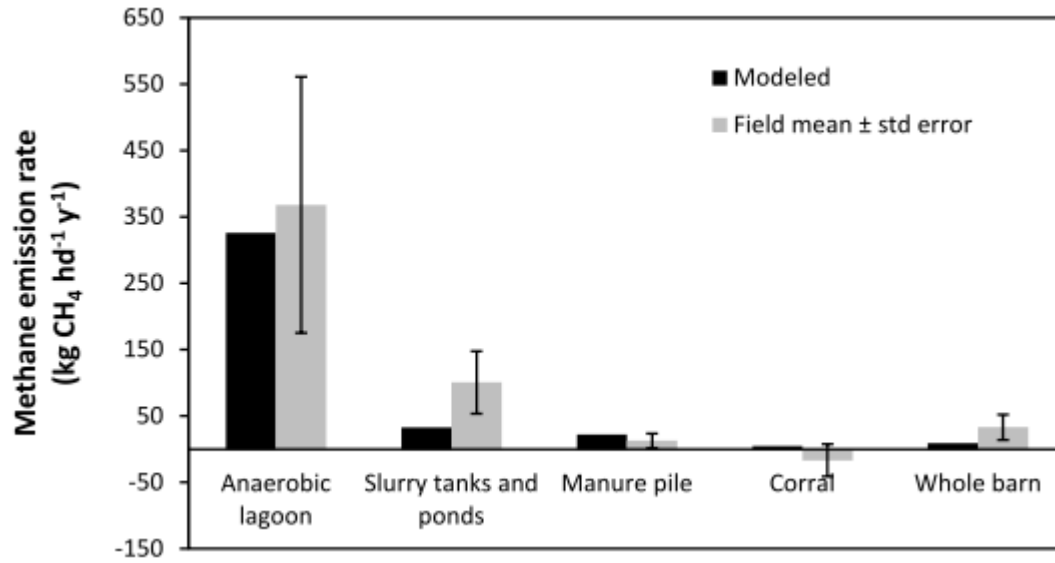
Minnesota	470000	290000	39	17	145	166961	1101942 ± 667844	0	145556 ± 72778
Mississippi	15000	7000	10	12	135	1236	8160 ± 4946	0	2967 ± 1484
Missouri	95000	50000	42	11	131	32560	214898 ± 130241	0	17055 ± 8528
Montana	14000	7000	19	42	150	2425	16007 ± 9701	0	10722 ± 5361
Nebraska	58000	20000	26	29	150	13233	87337 ± 52932	0	29520 ± 14760
Nevada	28000	10000	10	65	157	2569	16958 ± 10277	0	33402 ± 16701
New Hampshire	15000	7500	19	10	149	2583	17050 ± 10333	0	2719 ± 1360
New Jersey	7500	4000	25	6	138	1603	10581 ± 6412	0	769 ± 385
New Mexico	322000	140000	9	61	164	28173	185944 ± 112693	0	381905 ± 190953
New York	610000	330000	17	13	151	96041	633869 ± 384163	0	146886 ± 73443
North Carolina	44000	21000	11	10	152	4440	29306 ± 17761	0	8073 ± 4037
North Dakota	20000	10000	38	15	142	6604	43590 ± 26418	0	5214 ± 2607
Ohio	270000	125000	38	15	145	89967	593780 ± 359867	0	71026 ± 35513
Oklahoma	54000	20000	21	45	140	9409	62100 ± 37636	0	40325 ± 20162
Oregon	121000	65000	11	50	150	12245	80815 ± 48979	0	111315 ± 55658
Pennsylvania	543000	310000	24	6	147	118770	783881 ± 475079	0	59385 ± 29692
Rhode Island	1100	500	25	5	141	235	1549 ± 939	0	94 ± 47
South Carolina	16000	7000	8	18	144	1108	7315 ± 4433	0	4988 ± 2494
South Dakota	90000	25000	24	31	150	18635	122989 ± 74539	0	48140 ± 24070
Tennessee	50000	35000	12	4	140	5415	35737 ± 21659	0	3610 ± 1805
Texas	425000	230000	11	58	155	44298	292368 ± 177193	0	467144 ± 233572
Utah	87000	42000	15	56	151	11918	78662 ± 47674	0	88991 ± 44496
Vermont	135000	61000	17	13	145	20068	132448 ± 80271	0	30692 ± 15346
Virginia	95000	49000	11	5	145	9285	61282 ± 37140	0	8441 ± 4220
Washington	252000	122000	11	56	160	26658	175941 ± 106631	0	271425 ± 135712
West Virginia	10000	5000	23	7	134	1900	12543 ± 7602	0	1157 ± 578
Wisconsin	1265000	700000	38	17	150	443940	2930005 ± 1775761	0	397210 ± 198605
Wyoming	6000	5000	19	43	150	1123	7413 ± 4493	0	5084 ± 2542
<i>total kg N₂O y⁻¹</i>						<i>1709175</i>	<i>11280554 ± 6836699</i>	<i>0</i>	<i>6023128 ± 3011564</i>
total Tg CO₂e y⁻¹						0.51	3.36 ± 2.04	0	1.79 ± 0.90

738 **Figure 1.** Sources of N₂O and CH₄ on dairies. Thin arrows indicate movement of manure
739 between locations. Thick arrows indicate relative emission rate. Hardstandings are not shown
740 but have negligible emissions.



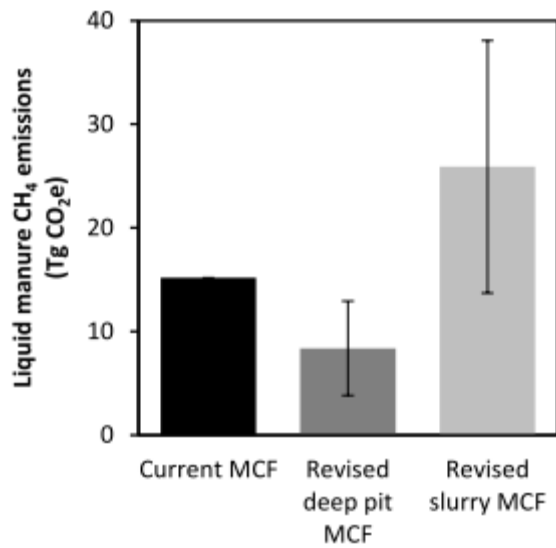
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746 **Figure 3.** Comparison of modeled CH₄ emissions and field measurement means and standard
747 errors for the largest CH₄ sources.



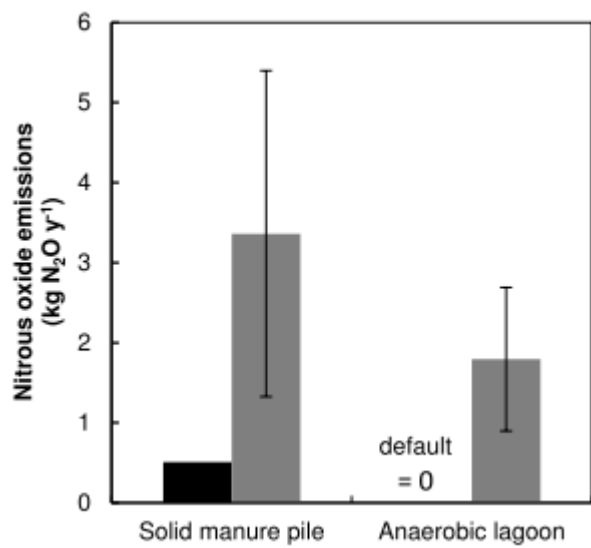
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750 **Figure 4.** Comparison of modeled CH₄ emissions for slurry using the current slurry *MCF* used
751 by the European Union and the revised *MCFs* for deep pit storage and slurry storage calculated
752 in this study (Table 4).



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754

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



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