

Greenhouse Gas Mitigation Opportunities in California Agriculture

Review of Emissions and Mitigation Potential of Animal Manure Management and Land Application of Manure

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ABSTRACT

Manure management, primarily in anaerobic lagoons on dairies, is estimated to be the largest source of greenhouse gases from California agriculture. However, no field measurements from dairies in California have been published. A review of the broader literature revealed that emissions from anaerobic lagoons had more than 10 times the global warming potential of emissions from solid manure piles. Capping anaerobic lagoons and flaring the emitted methane, or fully converting to anaerobic digesters, could reduce total methane emissions by 92% (~7.7 Tg CO₂e). Manure from farms is eventually applied to agricultural fields as fertilizer, where nitrous oxide is the primary greenhouse gas (GHG) emitted. Limited data are available on GHG emissions from manure-amended fields, and only two studies were conducted in California. This research suggested that fertilizing agricultural fields with manure rather than synthetic fertilizers results in lower GHG emissions as well as increased soil carbon storage. Despite the significance of dairy manure in GHG budgets at the state, regional, and global scale, this review reveals a surprising lack of field-scale research necessary to inform the development of best practices in California. Key areas of research for California include measurements of GHG emissions from dairy manure management systems and comparisons of GHG emissions from agricultural fields under different management practices.

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INTRODUCTION

In California, greenhouse gas (GHG) emissions from agriculture account for approximately 7% of total GHG emissions from all sources in the state (CARB 2011). Emissions from livestock manure management are estimated to be the greatest contributor, accounting for one-third of agricultural emissions (Table 1; CARB 2011). California Air Resources Board (CARB) estimates indicate that dairy manure management is by far the largest source of GHG emissions from livestock manure management in California, accounting for over 97% of the methane (CH₄) produced (Table 2; CARB 2011).

Table 1. Greenhouse Gas Emissions from California Agriculture in Carbon Dioxide Equivalents (CO₂e)

Agricultural source	2009 emissions (Tg CO ₂ e)
Manure management	10.34
Enteric fermentation	9.28
Soil management	9.02
Energy use	2.63
Rice cultivation	0.58
Histosol cultivation	0.16
Residue burning	0.06
Total	32.07

Source: CARB 2011.

Note: Emissions from manure spread on agricultural fields are included in soil management rather than manure management.

Table 2. Estimated GHG Emissions for 2010 from Manure Management in Different Livestock Industries in California

	Methane emissions (Tg CO ₂ e)	Nitrous oxide emissions (Tg CO ₂ e)
Dairy cattle	8.40037	1.12594
Beef cattle	0.02571	0.28108
Poultry	0.10508	0.04967
Swine	0.03588	0.00284
Goats	0.00008	0.00121
Horses	0.00398	0.02399
Sheep	0.00281	0.02354
Total	8.57392	1.50826

Source: CARB 2011.

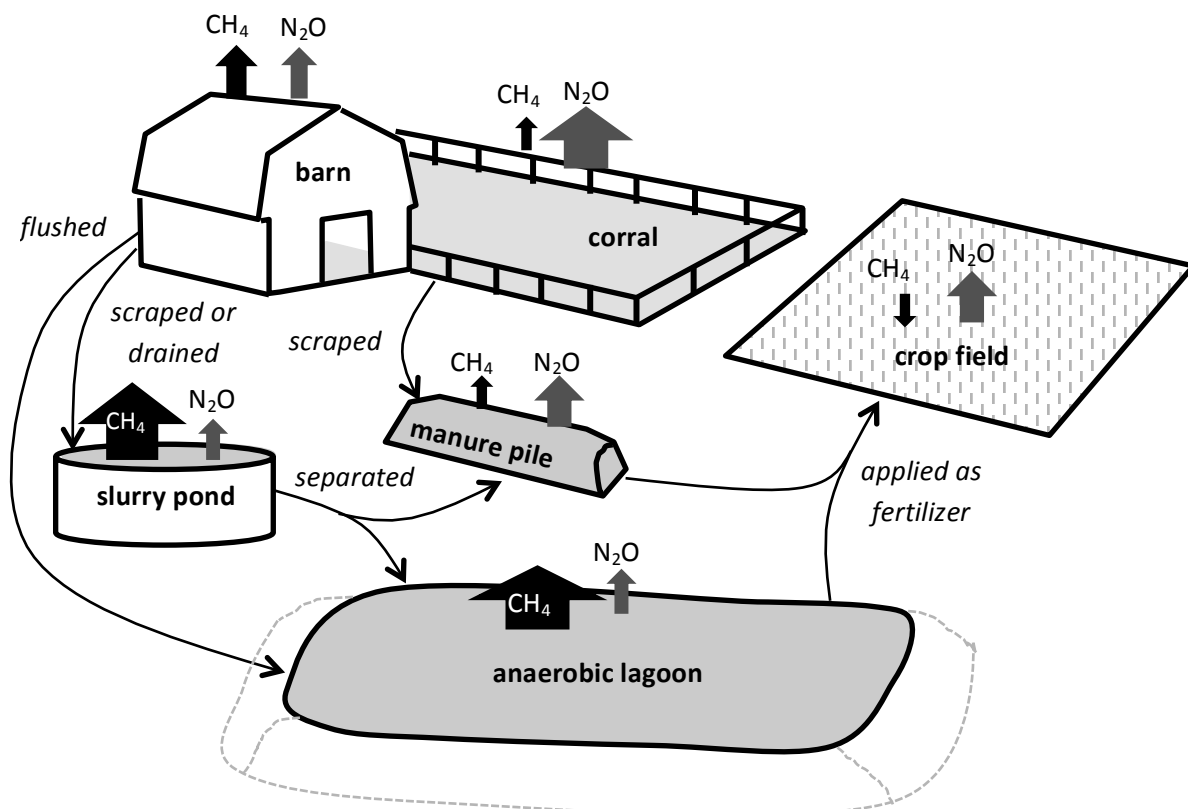
Note: Manure management does not include manure deposited or applied to pasture or agricultural fields.

Many areas on dairies are potential sources of greenhouse gases (Figure 1, USEPA 2011a; Meyer et al. 2011). The abundant carbon (C), nitrogen (N), and water in liquid and solid livestock waste promote microbial activity that drives the production of carbon dioxide (CO₂), CH₄, and nitrous oxide (N₂O). Carbon dioxide is primarily produced through aerobic microbial respiration associated with decomposition. Most C accounting schemes use aerobic decomposition as a baseline, thus C losses through aerobic respiration as CO₂ are not counted as part of the global warming potential. Methane is produced when the organic matter (including manure) is decomposed anaerobically by methanogens. Net CH₄ emissions are promoted under warmer temperatures, high N availability, and low oxygen

availability—conditions typical of manure storage areas. Methane is oxidized by aerobic and anaerobic methanotrophs. Methane consumption is common at anaerobic-aerobic interfaces, and upland soils are generally a net sink for CH_4 (Conrad 2007).

When organic matter is decomposed, organic N is mineralized into plant available forms such as ammonium (NH_4^+ , a potential air pollution source when it volatilizes as NH_3 and nitrate (NO_3^- , a potential water pollutant when it leaches out of the soil). Nitrous oxide production occurs as an intermediate step during two microbial processes: nitrification, the aerobic oxidation of NH_4^+ to NO_3^- , and denitrification, the anaerobic reduction of NO_3^- to dinitrogen gas (N_2 , the inert gas comprising the bulk of Earth's atmosphere; Firestone and Davidson 1989). Nitrous oxide production is greatest when aerobic and anaerobic conditions occur in close proximity, as in manure piles and corral soils.

Figure 1. Sources of N_2O and CH_4 on Dairies



Note: Thin arrows indicate movement of manure between locations. Block arrows indicate CH_4 and N_2O fluxes; thicker arrows indicate relatively greater emission rates.

Land application of manure directly from cows onto pasture or rangeland, or application to crop fields following on-dairy management, can be a significant source of greenhouse gases but may also contribute to soil C sequestration (DeLonge et al. 2013). Land-applied manure in California contained an estimated 231 Gg (1 Gg = 10^9 g) of N in 2009 and resulted in direct N_2O emissions of ~4 Gg, using the IPCC (2006) emissions factor of 0.01 g $\text{N}_2\text{O-N}$ g manure- N^{-1} , equivalent to 1.083 Tg CO_2e (1 Tg = 10^{12} g; CARB 2011). Methane fluxes on agricultural fields are not considered by regional and countrywide GHG accounting methods except for rice cultivation (IPCC 2006; USEPA 2011a), so no estimates are available for the statewide CH_4 emissions associated with land application of manure in California.

Reductions in GHG emissions from California agriculture, particularly dairy manure management, can contribute to the state's climate change mitigation goals. Following the approach of the IPCC, this report considers separately the GHG emissions associated with on-farm manure management and those associated with manure application to agricultural fields and rangelands. For each, this report summarizes current estimates and field measurements, identifies the greatest sources of greenhouse gases, and evaluates the biophysical mitigation potential of alternative management practices.

MANURE EMISSIONS

Background and Current Estimates

Dairies' manure management depends on whether manure is handled as solid or liquid, where it was deposited on the farm, how it will be used, and regional preferences for storage. Cows deposit manure onto dairy surfaces in dry lots, hardstandings, barn floors, and loafing pens. The length of time the cows are present and the frequency with which the manure is removed (by scraping, flushing, or vacuuming) are important determinants of the GHG fluxes. Liquid manure is generally stored in lagoons, slurry tanks, or settling ponds. Lagoons are not stirred, and anaerobic conditions develop rapidly. Slurry tanks have a larger solids content than lagoons; some solids typically float on the surface and form a surface crust. The crust provides a substrate that spans anaerobic and aerobic environments wherein N₂O production and CH₄ oxidation can both occur (Petersen, Amon, and Gattinger 2005; Petersen and Sommer 2011). Solid manure is composed of the solids scraped from dairy surfaces, raked from loafing pens, or separated from slurry. It is generally stored in outdoor, uncovered piles, sometimes inside dry lot corrals. Solid manure piles are heterogeneous in composition and can contain both aerobic and anaerobic areas, depending on moisture content and management practices. Loafing pens are typically smaller sources of manure, because milk cows are in them only when they are not in milking parlors or barn stalls.

Publicly available information about dairy manure management practices in California is limited. The USEPA (2011a) estimated GHG emissions from dairies using data on manure management practices calculated in ERG (2000) and adjusting for changes in the dairy industry (growth and consolidation) over time (Table 3). The 2009 estimates by CARB (2011) of the proportions of manure in each management system are similar to those of the USEPA (Table 3). The estimated distribution of manure management in California was typical of the distribution in the southwest United States, which relies heavily on anaerobic lagoons. Survey results of Meyer et al. (2011) are comparable, though they suggest the use of daily spread may be overestimated. Because Meyer et al. (2011) did not use the same categories as the EPA, the following conversions were used:

- The percent of dairies using solid storage was estimated as 100 minus the percent of dairies that did not separate solid and liquid manure.
- Liquid slurry use was calculated as equal to the percent of dairies that didn't separate solid and liquid manure.
- Anaerobic lagoon use was calculated as equal to the percent of dairies with storage ponds minus the liquid/slurry percent.
- Dry lot was estimated as equal to the percent of dairies where manure solids were piled.

The allocation between solid and liquid storage systems varies among dairies depending on infrastructure and management preferences and within a dairy depending on seasonal evaporation losses and management decisions (UC 2005).

Table 3. Manure Management Distribution (%) for Dairies and Heifer Facilities (Farms with No Milk Cows or Milking Facilities) in California and Glen and Tulare Counties

	Dairies					Dairy heifer facilities			
	Pasture	Daily spread	Solid storage	Liquid/slurry	Anaerobic lagoon	Daily spread	Dry lot	Liquid/slurry	Pasture
CA (USEPA)	1	11	9	20	59	11	88	1	1
CA (CARB)	0.93	10.8	9.24	20.1	56.5	10.8	88.3	0.87	0.93
Glen County	Na	0	63.2	36.8	52.7	0	80.1	na	na
Tulare County	Na	0	70.5	29.5	66.0	0	93.2	na	na

Sources: Glen and Tulare Counties: Meyer et al. (2011), California: USEPA (2011a) and CARB (2011).

Note: The USEPA (2011a) and CARB (2011) accountings were forced to add up to 100, whereas the survey by Meyer et al. (2011) allowed farmers to report all practices used and do not add up to 100. na = not available.

Based on the allocations in Table 3, CARB (2011) estimated GHG emissions by manure management system for dairy cows and dairy heifers using a Tier 2 approach (Table 4). The results reveal that the overwhelming majority of dairy cow manure GHG emissions are derived from liquid manure management and that most of these emissions are CH₄.

Table 4. Estimates of the Number of Cows Contributing Manure to Each Management System and the CH₄ and N₂O Emissions from That Manure Management in Tg CO₂e for 2009

Management practice	Head	CH ₄ (Tg CO ₂ e)	N ₂ O (Tg CO ₂ e)	Emissions per cow (g CO ₂ e/hd/d)
Dairy cows				
Anaerobic lagoon	1,039,082	7.323	0.348	20,227
Liquid/slurry	370,093	1.181	0.212	10,311
Deep pit	4,186	0.006	0.001	4,370
Anaerobic digester	40,786	0.033	0.007	2,685
Solid storage	170,073	0.061	0.096	2,528
Daily spread	198,755	0.009	0.014	321
Pasture	17,024	0.002	0	271
Dairy heifers				
Liquid/slurry	6,843	0.007	0.002	3,538
Dry lot	691,174	0.030	0.497	2,090
Daily spread	84,570	0.001	0.003	132
Pasture	7,244	0.000	0	122
<i>Total dairy cows</i>	<i>1,840,000</i>	<i>8.615</i>	<i>0.678</i>	<i>40,712</i>
<i>Total dairy heifers</i>	<i>782,920</i>	<i>0.039</i>	<i>0.502</i>	<i>5,882</i>

Source: CARB (2011).

Note: Nitrous oxide emissions from manure directly deposited on pasture are counted in "Agricultural Soil Management" and therefore are reported as 0 here.

Though useful, the data presented in Table 4 reflect two major uncertainties. First, manure is often managed in multiple ways for one cow (Figure 1; Meyer et al. 2011); how, if at all, CARB (2011) accounts for this fact is unclear. Second, the emissions estimates calculated by Tier 2 approaches are based on equations and emissions factors derived largely from laboratory incubations or pilot-scale studies conducted under a limited range of conditions. Thus, their applicability to farm-scale operations under a range of environmental conditions has not been confirmed. This research cannot address the first issue, but it explores the second by compiling field-scale emissions data and comparing it to Tier 2 calculations.

Approach to Data Compilation

Published field measurements of GHG emissions from on-dairy manure management on working and research dairies globally were reviewed and compared with emissions rates calculated using approaches and regional estimates from the IPCC (2006) and USEPA (2011a). Three broad categories of measurement areas were considered: (1) manure management (including liquid and solid systems), (2) holding area surfaces (dry lot corrals, loafing pens, hardstandings, and barn floors), and (3) whole barns. Following the approach of the IPCC (2006), liquid manure emissions data were divided into two groups: (1) anaerobic lagoons and (2) slurry tanks and settling ponds. Data from pilot and lab-scale studies were excluded. Also excluded were data from studies on beef cattle facilities, because their management and herd characteristics differ from those of dairies. Greenhouse gas emissions from anaerobic digesters are particularly difficult to quantify due to the variety of systems and operation practices (ex. retention times, effectiveness of seals in preventing gas escape, and composition of material entering the digester; Massé et al. 2011). Thus, the literature related to anaerobic digester emissions is not reviewed.

Thirty-two studies met the above criteria (Table 5). Most of these studies were conducted in North America and Europe (see Appendix Table 1A for data on location, measurement technique, and climate). No data were available for California, although results of at least two studies of GHG emissions from dairy sources in California are not yet published. Greenhouse gases were measured from the California dairy barn monitored for the National Air Emissions Monitoring Study (NAEMS) project (Zhao et al. 2010; F. Mitloehner, personal communication). Additionally, an on-farm study is measuring GHG emissions from a range of dairy sources to determine mechanisms controlling fluxes (Owen and Silver).

Methane flux measurements that included enteric fermentation by cows in barns, corrals, or whole dairies were corrected by subtracting $100 \text{ kg CH}_4 \text{ hd}^{-1} \text{ y}^{-1}$ (hd = head) (Mitloehner, Sun, and Karlik 2009), a moderate estimate compared with other estimates (Kinsman et al. 1995; IPCC 2006). Enteric emissions are dependent on many highly variable cow and feed characteristics, but the uncertainty in enteric emissions rates is smaller than that in the emission rates from manure management, as shown below.

Most studies included measurements of one source at different times (e.g., seasonally), or measurements from areas in which some management component was different (e.g., mixed versus static manure piles, barns with different flooring and scraping mechanisms), or both. Each permutation was included in the compilation. The average emissions rate for a given dairy source was calculated first by averaging the emissions rates compiled from each paper (though separate sources measured in the same study, such as primary and secondary lagoons, were treated separately and not averaged together) and then by averaging those values. This method avoided weighting the value toward studies and management practices with comparatively more measurements.

Emissions rates are presented in the units of $\text{g trace gas hd}^{-1} \text{ d}^{-1}$ and $\text{kg trace gas ha}^{-1} \text{ d}^{-1}$ when possible. Because data were not always provided to enable conversions between units, the average emissions rates presented below may include different studies depending on the units in which the emissions could be

reported. Data are reported as g CH₄ and g N₂O rather than g CH₄-C and g N₂O-N. Some have argued that expressing emissions per kg of fat-corrected milk is a better way to capture the efficiency of dairy production (Place and Mitloehner 2010; O'Brien et al. 2012), but few studies provided the information necessary to calculate these values. Similarly, livestock or animal units, which are the number of animals normalized by the typical mass of the animal (500 or 600 kg is usually used for dairy cows), are sometimes used. Given the lack of consistency in conversions and lack of information for some studies, livestock units were converted to head count when necessary. Methane and N₂O have different global warming potentials once in the atmosphere. To compare the total global warming potential of the GHG emissions from the measured areas, N₂O and CH₄ emissions rates were converted to 100-year CO₂e by multiplying by 298 and 25, respectively (IPCC 2006) and summing the two.

Table 5. Studies Reporting GHG Emissions Measurements from Working or Research Dairies and the Areas Measured by Each

Study	Lagoon	Slurry tank	Manure pile	Compost area	Open lot	Concrete pens	Barn	Whole dairy
Borhan et al. (2011a)	x			x	x		x	
Leytem et al. (2011)	x			x	x			
Borhan et al. (2011b)	x				x		x	
Bjorneberg et al. (2009)	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)		x	x					
Hensen et al. (2006)		x						x
Kaharabata et al. (1998)		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			
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	
Samer et al. (2012)							x	

Snell et al. (2003)	x
van Vliet et al. (2004)	x
Zhang et al. (2005)	x
McGinn and Beauchemin (2012)	x

The IPCC (2006) Tier 2 approach, used with values from the IPCC (2006) and the USEPA (2011a) and equations 1 and 2 below, was applied to predict CH₄ and N₂O emissions rates for each source. The resulting values were then compared with the average field measurements. The IPCC Tier 2 method calculates CH₄ emissions (EF, g CH₄ hd⁻¹ y⁻¹) on the basis of the volatile solid content of manure excreted by cows (VS, kg VS hd⁻¹ y⁻¹), a CH₄ conversion factor (MCF, %) based on the manure management practice, and the maximum possible CH₄ production rate from the volatile solids in the manure (B_o, m³ CH₄ kg VS⁻¹)(IPCC 2006):

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

The IPCC Tier 2 method calculates direct N₂O emissions on the basis of an emissions factor (EF_{N_2O} , kg N₂O-N kg N excreted⁻¹) and annual N excretion rates, which are a function of cows' energy intake, the percent of crude protein, the milk production rate and protein content, cow growth, typical animal mass, and (IPCC equations 10.31, 10.32, and 10.33; IPCC 2006). EF_{N_2O} can be converted into N₂O emissions rates equivalent to those measured here (N_{2O_D} , g N₂O hd⁻¹ d⁻¹, where the subscript D refers to direct emissions) using the typical animal mass (TAM , kg) and country- or region-specific N excretion rates (N_{ex} , kg N 1000 kg TAM^{-1} d⁻¹):

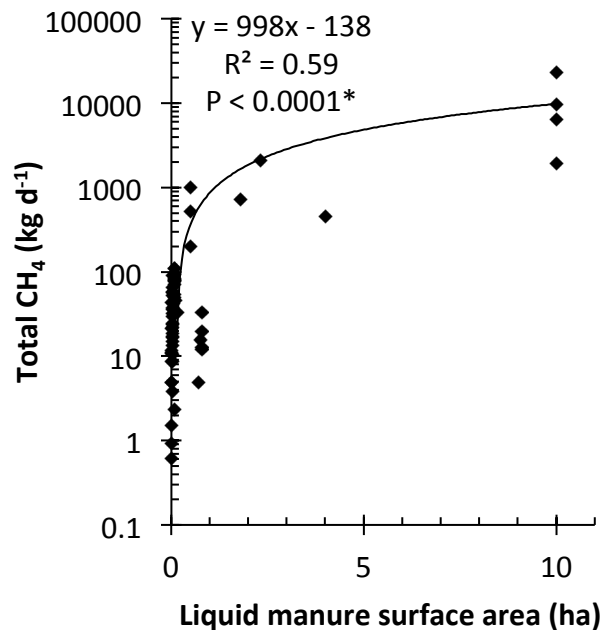
$$N_{2O_D} = EF_{N_2O} \times 44/28 \times TAM \times N_{ex} \quad (\text{eq. 2}).$$

Indirect N₂O emissions, derived from the oxidation of gaseous emissions such as ammonia (NH₃) and nitrous oxides (NO_x), are important for calculating the amount of N remaining in manure for its use as an organic fertilizer (IPCC 2006). No indirect N₂O emissions are included in the calculations here; values reported here should be considered minimum values. The complete data compilation is available in appendix tables A2–8.

Emissions from Manure Storage Systems

Consistent with IPCC and EPA estimates, the greatest emissions on dairies were measured from manure storage. Total emissions from liquid systems averaged ~19,000 g CO₂e hd⁻¹ d⁻¹, and those from solid systems averaged ~1,700 g CO₂e hd⁻¹ d⁻¹ (Table 6). Liquid manure systems had greater CH₄ emissions than solid systems and smaller, but non-zero, N₂O emissions (Table 6). By contrast, IPCC emissions factors predicted zero N₂O emissions from liquid manure except for slurry systems with a crust (IPCC 2006). The range of GHG emissions from anaerobic lagoons and slurry tanks were similar, although anaerobic lagoons tended to emit more CH₄ on a per head basis than slurry tanks (Table 6). Total CH₄ emissions (kg d⁻¹) from liquid manure systems were positively correlated with surface area ($r^2 = 0.59$, $P < 0.0001$, Figure 2). Despite the importance of manure composition in emissions calculations, few studies reported volatile solid content or other measures (tables A2 and A3).

Figure 2. Total Daily CH₄ Emissions from Liquid Manure Management Increased with Surface Area



Tier 2-predicted CH₄ emissions from anaerobic lagoons were 892 g hd⁻¹ d⁻¹, < 20% below and within the standard error of the field measurement mean of 1097 ± 591 g hd⁻¹ d⁻¹ (Table 7). Slurry tank and pond CH₄ emissions were underestimated by the predicted rate of 81 g hd⁻¹ d⁻¹, which is less than a third of the field mean of 276 ± 129 g hd⁻¹ d⁻¹ (Table 7). The predicted CH₄ emissions rates may not adequately account for the emissions related to field-scale events such as periodic tank emptying and surface crust disturbances. Crust disturbances by rainfall increased CH₄ emissions by 1.2–4 times in one study (Kaharabata et al. 1998), whereas crust thawing increased CH₄ emissions over an order of magnitude (VanderZaag et al. 2011). These episodic releases of CH₄ comprise a large portion of the total CH₄ emissions but are not easily included in emission factors based on environmental characteristics or region. Additionally, slurry remaining in the tank after emptying inoculates fresh slurry with microbes, generally increasing CH₄ emissions and decreasing the time to the onset of CH₄ production once the tank is refilled (Zeeman 1991), although this effect may be less important in cold climates (VanderZaag et al. 2011). Volatile solid carryover is considered in the USEPA default emissions factors (Mangino et al. 2002; USEPA 2011a) but not in the IPCC emissions factors.

Non-zero, direct N₂O emissions were always detected from anaerobic lagoons, with a mean of 1.4 ± 1.2 g hd⁻¹ d⁻¹ (tables 6 and A2), and thus the IPCC N₂O emissions factor of zero was an underestimate (Table 8). Scaling these emissions to anaerobic lagoon use in California in 2009 (Table 4) suggests *additional* N₂O emissions from anaerobic lagoons of 0.16±0.14 Tg CO₂e, or a mean increase of nearly 50% over the CARB (2011) estimate. Tier 2-predicted slurry tank N₂O emissions of 2.1 g hd⁻¹ d⁻¹ were similar to the field mean of 0.7 ± 0.7 g hd⁻¹ d⁻¹ when the emissions factor for slurry with a natural crust was used, as was the case for most of the slurry tanks (tables 8 and A3).

The comparison between the Tier 2 values and the field measurement means suggests that anaerobic lagoon emissions are reasonably well predicted and that estimates for California are likely to be reliable

(tables 7 and 8). Slurry tank emissions estimates appear less reliable and suggest the California value may be an underestimate.

Manure pile emissions were measured in nine studies, and a large manure composting area was measured in one (tables A1 and A4). Few studies reported manure characteristics, but of those that did, the dry matter (DM) content was 19–45%, manure N content was 0.11–3.92 g N kg DM⁻¹, manure C content was 4.8–65 g C kg DM⁻¹, and the volatile solid content of manure was 16–93% (Table A4). Although the sample size was low, these ranges reveal the variety and heterogeneity in chemical and physical characteristics of solid manure. Data from Gupta et al. (2007) were excluded from Table 6 because they were reported in units of mass trace gas per mass manure per time, without adequate information to confidently convert to the units used here. The estimates of mass gas hd⁻¹ d⁻¹ in Table A4 were calculated assuming 2.6 kg DM manure production hd⁻¹ d⁻¹, the IPCC (2006) default for volatile solid production by dairy cows in India.

Greenhouse gas emissions from manure piles spanned 3 to 4 orders of magnitude (Table 6), and much of this variation appeared to be related to measurement technique rather than variation among piles. Studies that used sample masts, tracer gas techniques, or large dynamic flux chambers measured higher emissions than studies using smaller dynamic or static chambers (Table A4). For example, CH₄ emissions measured using large footprint techniques were 93 ± 25 kg ha⁻¹ d⁻¹ (n = 5), or 104 g hd⁻¹ d⁻¹ (n = 1), compared with 0.091 kg ha⁻¹ d⁻¹ (n = 1), or 0.023 ± 0.012 g hd⁻¹ d⁻¹ (n = 3), from small chambers. The number of studies is extremely low, however, and more data are needed to confirm this observation. Theoretically, measurement methods that measure gas fluxes from larger areas likely have greater capacity to integrate the spatial and temporal variability of pile emissions arising from manure piles' varied composition and oxygen availability (Parkin and Kaspar 2004; Sommer et al. 2004).

Methane emissions were greatest when manure piles were first constructed. Data were inadequate to determine if subsequent additions to the piles also resulted in greater emissions. Mixed piles had greater CH₄ emissions than static ones, consistent with the findings of pilot-scale studies (Yamulki 2006), but contrary to expectations that mixing would aerate the pile and decrease CH₄ production. The predicted CH₄ emissions rate from piles was 6.0 g hd⁻¹ d⁻¹ (Table 7), which greatly underestimated the large-footprint measurements and overestimated the small chamber measurements. Thus, the California 2009 estimate (Table 7) may also be an underestimate.

Manure piles had high N₂O emissions, with a field measurement mean second only to corrals (Table 6). Ahn et al. (2011) found that N₂O emissions increased when the piles were mixed. The predicted N₂O emissions rate was 1.2 g hd⁻¹ d⁻¹, within the standard error of the field measurement mean (Table 8). The California 2009 estimate was higher, partially because it includes indirect N₂O emissions, which arise from the deposition of manure-derived NH₃ on soil, where it is converted to N₂O. Increasing pile aeration may decrease N₂O emissions, but more data are needed to confirm this hypothesis.

Table 6. Summary of the Means and Ranges of N₂O and CH₄ Emissions Rates Measured by the Studies in Table 1

	Mean ± standard error (n)						Global warming potential = 25 x CH ₄ + 298 x N ₂ O Mean ± standard error	
	(kg/ha/d)			(g/ha/d)			CO ₂ e (kg/ha/d)	CO ₂ e (g/ha/d)
	CO ₂	N ₂ O	CH ₄	CO ₂	N ₂ O	CH ₄		
Manure management*	3987 ± 1349 (12) 52 - 126308	4.5 ± 1.5 (13) 0.00 - 27	485 ± 116 (20) 0 - 2307	1929 ± 576 (10) 13 - 6575	1.7 ± 0.7 (10) 0 - 8322	619 ± 286 (18) 0 - 7710	13465 ± 3344	15985 ± 7380
Solid manure piles*	7334 ± 2816 (4) 367 - 12631	7.0 ± 2.4 (7) 0.13 - 27	78 ± 25 (6) 0.1 - 352	2066 ± 1904 (2) 162 - 9716	2.9 ± 1.8 (4) 0.06 - 21	35 ± 30 (4) 0 - 271	4039 ± 1332	1732 ± 1287
Liquid	1755 ± 943 (8) 52 - 8548	1.5 ± 0.9 (6) 0.027 - 10.3	660 ± 142 (14) 0 - 2307	1883 ± 728 (8) 13 - 6575	1.1 ± 0.6 (6) 0.010 - 6.8	745 ± 350 (14) 0 - 7710	16944 ± 3803	18935 ± 8952
Slurry tanks and ponds	nm	1.3 ± 1.1 (3) 0 - 19	756 ± 268 (6) 0 - 2027	nm	0.7 ± 0.7 (3) 0 - 12.3	276 ± 129 (6) 0 - 899	19280 ± 7045	7117 ± 3442
Anaerobic lagoons	1755 ± 943 (6) 52 - 8548	1.7 ± 1.6 (3) 0.027 - 8.5	588 ± 158 (8) 7.0 - 2307	1883 ± 728 (6) 13 - 6575	1.4 ± 1.2 (3) 0.010 - 6.5	1097 ± 591 (8) 10 - 7710	15207 ± 4421	27822 ± 15125
Surfaces	1349 ± 637 (7) 137 - 10007	0.50 ± 0.21 (9) 0.002 - 5.98	2.81 ± 9.47 (12) -90.1 - 118	5138 ± 3895 (7) 69 - 55594	1.87 ± 1.17 (9) 0 - 33.2	5.78 ± 26.9 (12) -222 - 653	215 ± 299	703 ± 1022
Corrals**	2263 ± 1411 (3) 324 - 10007	1.18 ± 0.35 (3) 0.033 - 5.98	-9.5 ± 21 (5) -90.1 - 118	11623 ± 8329 (3) 366 - 55594	5.5 ± 2.5 (3) 0.2 - 33	0.4 ± 68 (5) -222 - 653	115 ± 639	1658 ± 2464
Hard-standings**	nm	0.009 ± 0.002 (2) 0.002 - 0.030	20 ± 17 (3) 0.05 - 59	nm	0.0012 ± 0.0002 (2) 0.0002 - 0.003	19 ± 2.1 (3) 0.01 - 67	493 ± 428	481 ± 54
Barn floor	583 ± 263 (4) 137 - 1596	0.23 ± 0.16 (4) 0.01 - 1.0	5.6 ± 4.3 (4) 0.4 - 28	257 ± 108 (4) 69 - 684	0.07 ± 0.04 (4) 0.004 - 0.3	2.4 ± 1.8 (4) 0.1 - 12	209 ± 153	81 ± 56
Whole barn**	21206 ± 15824 (3) 1060 - 101738	33 ± 30 (2) 0 - 63	158 ± 80 (7) -80 - 887	19737 ± 15087 (3) 747 - 96048	31 ± 29 (2) 0 - 61	153 ± 75 (7) -70 - 837	13657 ± 10995	13170 ± 10524
Whole dairy**	nm	nm	nm	nm	nm	309 ± 96 (18) -203 - 1005	nm	7735 ± 2405

*Excludes data from Gupta et al. (2007), which could not with confidence be converted into the units used here.

**Where necessary, CH₄ emissions rates have been corrected for enteric emissions assuming 100 kg hd⁻¹ y⁻¹. Carbon dioxide emissions were not corrected for respiration.

Note: The mean for a measurement area was calculated by first averaging measurements within studies and then averaging across studies (n = number of studies). The exception was whole dairy measurements, which were treated individually and not grouped by study (i.e., n = number of dairies measured). In this table, values for the manure management row are the average of solid and liquid manure management systems, values for the liquid row are the average of all liquid systems (slurry tanks, ponds, and anaerobic lagoons), and values for the surfaces row are the average of corrals, hard-standings, and barn floors. nm = not measured.

Table 7. Calculation of Predicted CH₄ Emissions Rates Using the Tier 2 Approach (eq. 1) and Comparison with Field-Measured Values (Mean ± Standard Error (n)) and the 2009 Estimates for California from CARB (2011)

	VS (kg/hd /y)	VS source	MCF (%)	MCF source	B ₀ (m ³ /k g)	B ₀ source	CH ₄ emissions rate (g/hd/d)		
							Tier 2 predicted	Field measured	California estimate
Anaerobic lagoons	2770	USEPA (2011a), average ID, NM, TX, NC	74	USEPA (2011a), average ID, NM, TX, NC	0.24	IPCC (2006)/USEPA (2011a)	892	1097 ± 591 (8)	772
Slurry tanks and ponds	1861.5	IPCC (2006), Western Europe	10	IPCC (2006), MAT<10, with crust	0.24	IPCC (2006), Western Europe	81	276 ± 129 (6)	350
Manure pile	2750	USEPA (2011a), average TX and ID	0.5	USEPA (2011)	0.24	USEPA (2011a)	6.0	35 ± 30 (4)	39
Corral	2800	USEPA (2011a)	1	IPCC (2006)/USEPA (2011a), cool MAT	0.24	IPCC (2006)/USEPA (2011a)	12.2	0.4 ± 68 (5)	4.8
Concrete hardstanding	1861.5	IPCC (2006), Western Europe	1	IPCC (2006), cool MAT	0.24	IPCC (2006), Western Europe	8.1	3 ± 3 (2)	NA
Brick hardstanding	1022	IPCC (2006), Asia	1	IPCC (2006), cool MAT	0.13	IPCC (2006), Asia	2.4	52 ± 21 (1)	NA
Whole barn	1861.5	IPCC (2006), Western Europe	3	IPCC (2006), cool MAT	0.24	IPCC (2006), Western Europe	24	153 ± 75 (7)	158
Whole dairy (= corral + manure pile + slurry tank + whole barn)							117	309 ± 96 (18)	816

Note: VS = volatile solids, MCF = methane conversion factor, B₀ = the maximum possible CH₄ production rate from the volatile solids in the manure, NA = not available. The 2009 California deep pit estimate was used as the whole barn value, and the whole dairy value was the sum of anaerobic lagoons, corrals, and manure piles.

Table 8. Calculation of N₂O Emissions Rates Using the Tier 2 Approach (Eq. 2) and Comparison with Field-Measured Values (Mean ± Standard Error (n)) and the 2009 Estimates for California from CARB (2011)

	EF _{N₂O} (kg N ₂ O-N/ kg N excreted)	TAM (kg/hd)	N _{ex} (kg N/ 1000kg TAM/d)	EF _{N₂O} and N _{ex} sources	N ₂ O emissions rate (g/hd/d)		
					Tier 2 predicted	Field measured	California estimate
Anaerobic lagoon	0	600	0.25	USEPA (2011a), average of ID and TX	0	1.4 ± 1.2 (3)	3.1
Slurry tanks and ponds	0.005	600	0.44	IPCC (2006), North America, with crust	2.1	0.7 ± 0.7 (3)	5.3
Solid manure pile	0.005	600	0.26	USEPA (2011a), ID	1.2	2.9 ± 1.8 (4)	5.2
Corral	0.02	600	0.25	USEPA (2011a), average of ID and TX	4.8	5.5 ± 2.5 (3)	6.6
Concrete hardstanding	0.02	600	0.44	IPCC (2006)	8.3	0.0012 ± 0.0002 (2)	NA
Whole barn	0.002	600	0.48	IPCC (2006), western Europe	0.91	31 ± 29 (2)	1.4

Note: EF_{N₂O} = emissions factor, TAM = typical animal mass, N_{ex} = country- or region-specific N excretion rates, NA = not available. CARB (2011) estimates for liquid systems include indirect N₂O emissions, which account for the non-zero value for anaerobic lagoons. The 2009 California deep pit estimate was used as the whole barn value.

Emissions from Dairy Surfaces

Greenhouse gas emissions from dairy surfaces depended on whether the surface was soil (corrals), soil covered with bedding (loafing pens), or paved (hardstandings and barn floors) and on the frequency with which accumulated manure was removed. The spatial and temporal heterogeneity in moisture content of the soil surfaces contributed to a wide range in CH₄ emissions, including positive and negative fluxes, and some of the highest N₂O emissions of any source area (tables 6 and A5). Emissions from hardstandings and barn floors were much lower (Table 6); N₂O emissions were near zero.

In corrals and loafing pens, CH₄ fluxes ranged over several orders of magnitude; the mean was 0.41 ± 68 g hd⁻¹ d⁻¹ or -9.5 ± 21 kg ha⁻¹ d⁻¹, where the negative sign indicates CH₄ uptake (Table 6). The mean values had different signs due to the difference in the studies included in the calculation (i.e., emissions measurements available in g hd⁻¹ d⁻¹ or kg ha⁻¹ d⁻¹). Methane emissions measured from sandy and sandy loam soils using large-footprint techniques [sample mast by Leytem et al. (2011); open path Fourier transform infrared spectroscopy (FTIR) by Bjorneberg et al. (2009), and SF₆ tracer by Kaharabata et al. (2000)] were one to two orders of magnitude greater than emissions measured from loam to clay loam soils using dynamic chambers (Borhan et al. 2011a, 2011b). Methane can be readily oxidized in well-drained soil (Conrad 2007), thus sandy soils are generally thought to have lower CH₄ emissions potential than more finely textured soils. Pen conditions such as manure accumulation and high moisture, however, can slow gas diffusion and favor methanogenesis over CH₄ oxidation. The large- and small-footprint measurement techniques had different ranges (-222 – 653 g hd⁻¹ d⁻¹ and 5 – 22 g hd⁻¹ d⁻¹, respectively; Table A5), but the means were of the same order of magnitude (-6.1 ± 124 g hd⁻¹ d⁻¹ and 10.2 ± 5.8 g hd⁻¹ d⁻¹, respectively). As with manure management systems, small chambers deployed for shorter time periods may miss hotspots (both sources and sinks) that are better integrated into large-footprint measurements.

Corral CH₄ emissions were negative in 4 of 17 cases. Negative fluxes occurred in late summer when the soils were dry or in winter when the soils were cold or frozen (Table A5). Methane emissions were not correlated with either temperature or precipitation. Manure accumulation in the corrals was likely one of the most important factors driving positive CH₄ fluxes. Borhan et al. (2011b) measured greater CH₄ emissions from a dry lot corral holding heifers than from loafing pens measured at the same time (11 vs. 3.1 g hd⁻¹ d⁻¹, respectively; Table A5). The open lots likely had a greater influx of relatively fresh organic matter and localized high-moisture urine patches, facilitating production of anaerobic conditions and methanogenesis.

Nitrous oxide emissions from corrals were measured by three studies (Borhan et al. 2011a, 2011b; Leytem et al. 2011) and ranged over three orders of magnitude (Table 6). Corrals were the greatest source of N₂O considered on a per-head basis with a mean of 5.53 ± 2.54 g hd⁻¹ d⁻¹ (Table 6). The highest emissions were measured in late spring, when a combination of warmer temperatures and moist soils promoted N₂O production (Table A5). The lowest values were < 50% of the highest emissions and occurred in late fall and winter (Table A5). As in the case of CH₄ emissions, the loafing pen measured by Borhan et al. (2011b) had lower N₂O emissions rates than those of the corral (0.53 vs. 0.73 kg ha⁻¹ d⁻¹ or 0.60 vs. 2.41 g hd⁻¹ d⁻¹, respectively), likely for the same reasons discussed above (less manure accumulation and drier conditions in the loafing pen than in the corral). Methane and N₂O fluxes from corrals were weakly positively correlated ($r^2 = 0.28$, $P < 0.05$) and suggest that at least a portion of the N₂O fluxes were derived from denitrification, which requires the same general environmental conditions (warm temperatures, abundant labile C, anaerobic conditions) as methanogenesis.

Tier 2-predicted CH₄ and N₂O emissions from corrals were within the standard errors of the field measurement means (Tables 7 and 8). Predicted CH₄ emissions were 12.2 g hd⁻¹ d⁻¹ (Table 7), close to the mean of 0.41 ± 68 g hd⁻¹ d⁻¹ but not very informative given the large range of measurements (Table 6). Similarly, the California 2009 estimate was of the same order of magnitude as the Tier 2 estimate and

within the range of field-measured emissions rates. Even the largest field measurements were an order of magnitude smaller than the emissions from liquid manure and thus did not affect assessment of the greatest sources of CH₄ on California dairies.

The predicted N₂O emissions rate was 4.8 g hd⁻¹ d⁻¹, close to the field measurement mean of 5.5 ± 2.5 g hd⁻¹ d⁻¹ (Table 8). The N₂O data were limited to three studies: one in Idaho (Leytem et al. 2011) and two in Texas (Borhan et al. 2011a, 2011b) (Table A5). The 2009 California estimate (Table 8) was similar to the other values, suggesting that the corrals may be the largest N₂O emissions source on dairies. However, options to minimize N₂O emissions from corrals are limited, and those that are feasible, such as minimizing standing water, are generally already in use.

Hardstandings and barn floors were concrete or paved with the exceptions of the freestall bedding in an open-sided barn measured by Borhan et al. (2011a, 2011b) and the brick corrals measured by Gao et al. (2011) (Table A6). Frequent scraping of the paved and concrete surfaces likely limited substrate availability and the occurrence of anaerobic conditions; thus CH₄ and N₂O emissions were orders of magnitude lower than from other dairy sources (Table 6).

Methane emissions from most surfaces were < 8 g hd⁻¹ d⁻¹ with the exception of the brick hardstanding, which emitted 37 and 67 g hd⁻¹ d⁻¹ in the winter and summer, respectively (Gao et al. 2011) (Table A6). The higher CH₄ emissions from the brick hardstanding were likely due to less frequent scraping in this study (every 1–4 weeks vs. daily for most others, Table A6; Gao et al. 2011). Consistent with this finding, Adviento-Borbe et al. (2010) observed a significant, positive correlation between CH₄ emissions and manure depth on the barn floor. Hardstanding emissions did not have strong seasonal differences, likely due to the rapid drying of slurry in the summer, which limited anaerobic conditions, and to cold temperatures in the winter (Ellis et al. 2001). The Ellis et al. (2001) and Adviento-Borbe et al. (2010) studies indicate that solid surfaces that are scraped frequently are negligible sources of CH₄ and that manure accumulation is the primary factor affecting CH₄ emissions from hard dairy surfaces. Nitrous oxide emissions from hardstandings and barn floors were near zero; the maximum of 0.3 g hd⁻¹ d⁻¹ is from freestall bedding (Table A6; Borhan et al. 2011a). Daily scraping and dry bedding limited conditions conducive to nitrification, denitrification, or both.

IPCC default emissions factors for dry lots include paved and unpaved surfaces, such as the hardstandings and barn floors considered here. This study calculated the predicted CH₄ emissions rates separately for the brick hardstanding and the other surfaces to account for regional differences in dairy management. Predicted CH₄ emissions were 8.1 g hd⁻¹ d⁻¹ for the hardstandings and barn floors, the same order of magnitude as the field mean of 3 ± 3 g hd⁻¹ d⁻¹ (Table 6). Predicted CH₄ emissions from the brick hardstanding were even lower at 2.4 g/hd/y, less than 5% of the average from Gao et al. (2011) (Table 6). This result is likely due to use of the default Asian *B_o* and *V_S* values, which may not represent the modern Holstein dairy that was measured (Gao et al. 2011). Thus, the IPCC Tier 2 approach is adequate for calculating CH₄ emissions from most dairy surfaces, but it may need to be refined for the rapidly modernizing Asian dairy industry.

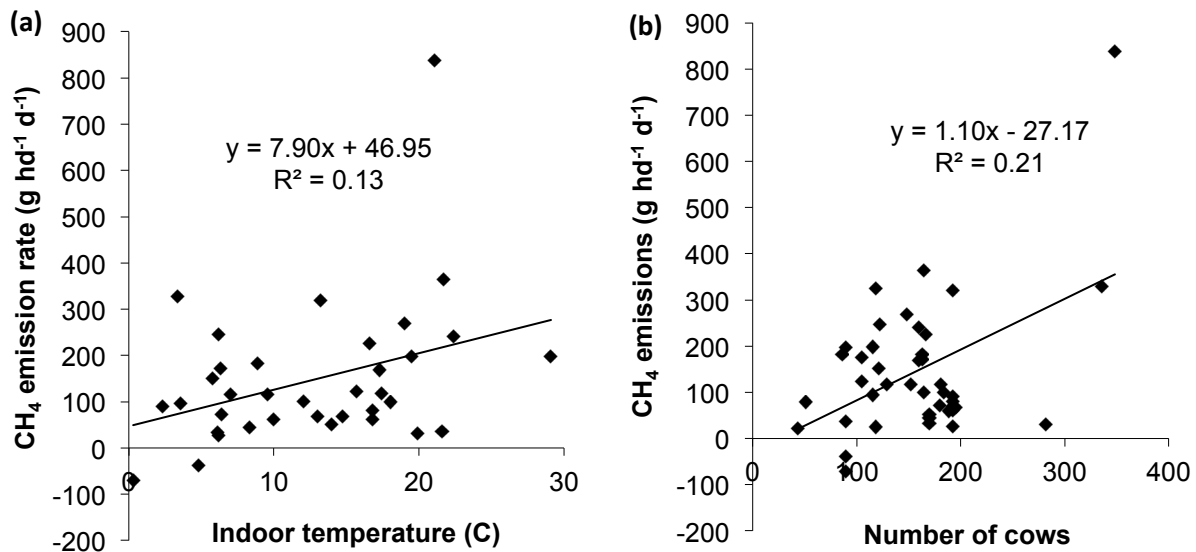
Predicted N₂O emissions for the hardstandings and barn floor were the same as for the dry lot at 8.3 g hd⁻¹ d⁻¹, three orders of magnitude larger than the field mean of 0.0012 ± 0.0002 g hd⁻¹ d⁻¹ (Table 7). Hardstandings and barn floors are not the primary manure management area for most dairies, and therefore these emissions are often excluded from local and regional estimates. More measurements of N₂O emissions for hardstandings and barn floors are needed to determine if manure accumulation might increase N₂O emissions to rates comparable with those predicted by the Tier 2 approach.

In summary, GHG emissions from dairy surfaces varied widely depending on the surface composition and the amount of manure accumulation on that surface. Dry lot corrals can be significant sources of N₂O. Low moisture content, low temperature, and frequent manure removal appeared to limit GHG emissions from dairy surfaces.

Emissions from Whole Barns

Six studies measured CH₄ emissions from whole barns and two measured N₂O (Table A7). Kinsman et al. (1995) was the only study that measured an empty barn, i.e., excluded enteric emissions. All the measured barns were enclosed tiestall or freestall barns, unlike most barns in California which are typically open-sided. Mean CH₄ emissions (corrected for enteric emissions) were $158 \pm 80 \text{ kg ha}^{-1} \text{ d}^{-1}$ or $153 \pm 75 \text{ g hd}^{-1} \text{ d}^{-1}$, intermediate between emissions from manure management and from surfaces (Table 5). Samer et al. (2012) measured the highest barn CH₄ emissions in summer from a barn that had the highest ventilation rate, highest number of cows, and largest cows (Table A7). The lowest CH₄ emissions were measured in winter from a barn with the fewest cows and a relatively high ventilation rate (Table A7) (Marik and Levin 1996). Methane emissions were usually higher in summer than in winter and were weakly positively correlated with indoor temperature (Figure 3a). Methane emissions also increased with number of cows (Figure 3b), which could be due to the greater manure accumulation associated with greater numbers of cows, or it could indicate that this study's enteric emissions correction may be an underestimate for European dairies that make up the bulk of the studies considered here. Zhang et al. (2005) observed higher CH₄ emissions from barns with slatted floors than from ones with solid floors. This finding may reflect the manure handling systems associated with the different floor types, because almost all the slatted floor barns had flushing or circulation systems, whereas solid floors were mechanically scraped.

Figure 3. Whole Barn CH₄ Emissions (Corrected for Enteric Emissions Assuming 100 kg CH₄ hd⁻¹ y⁻¹) Increase with (a) Indoor Barn Temperature and (b) Number of Cows



Measurements of nitrous oxide emissions from whole barns in two studies yielded very different results. The mean of measurements by Samer et al. (2012) was $60.36 \pm 1.08 \text{ g hd}^{-1} \text{ d}^{-1}$ and was the largest calculated for any dairy source. In contrast, and despite using a similar measurement technique, the mean from Zhang et al. (2005) was about 25 times less ($2.37 \pm 0.47 \text{ g hd}^{-1} \text{ d}^{-1}$, Table A7) and was comparable to N₂O emissions from other dairy sources (Table 6). Nitrous oxide fluxes typically vary widely in space

and time as a result of the sensitivity of nitrification and denitrification to local conditions (Firestone and Davidson 1989). In this case, differences in barn conditions in the Samer et al. (2012) and Zhang et al. (2005) studies appeared to create large differences in emissions, but identifying the key factors driving these differences is difficult. Zhang et al. (2005) found that N₂O emissions were generally greater in winter than summer, and no specific combinations of flooring or manure handling appeared to consistently produce higher or lower values.

The IPCC has no specific emissions factors for barns, but deep bedding and pit storage in barns are included in manure management calculations. Though few of the barns considered here had deep bedding or pit storage, they collect some manure and so this study calculated emissions under the assumption that pit storage was present (Tables 6 and 7). Predicted CH₄ emissions were 24 g hd⁻¹ d⁻¹, much lower than the field mean. This discrepancy could result from an underestimation of enteric emissions in this study or an underestimation in the emissions factor of the amount of manure typically present in barns. After recalculation of the field mean CH₄ emissions using the IPCC (2006) default enteric emissions rate for Western Europe (320 g hd⁻¹ d⁻¹), the field mean decreased to 124 ± 80 g hd⁻¹ d⁻¹, still well above the predicted value. Thus, the data suggest CH₄ emissions from manure in barns may be as great as or greater than emissions from subfloor manure storage tanks or deep bedding, and emissions factors accounting for these sources should be developed and included in future emissions calculations.

The predicted N₂O emissions rate of 0.91 g hd⁻¹ d⁻¹ was two orders of magnitude less than the field mean and less than 40% of the mean of the measurements from Zhang et al. (2005). Enteric N₂O emissions are not considered by the IPCC (2006), but they are generally low (≤0.5 g hd⁻¹ d⁻¹; Mitloehner, Sun, and Karlik 2009) and are insufficient to account for the difference between measured and predicted values. This result suggests that the current emissions factors may underestimate N₂O emissions from some barns, but this possibility is highly uncertain, reflecting data from only two studies.

In California, deep pit storage is relatively rare, accounting for manure treatment of less than 5,000 hd (Table 3), so the accuracy of this estimate is not as critical as it is for other liquid systems. Additionally, most California barns are open-sided and are scraped or flushed. Thus, barns in California are likely small sources of GHG emissions, provided they do not have deep pit storage and manure is removed frequently (as discussed for paved dairy surfaces).

Emissions from Whole Dairies

Comparatively fast and non-intrusive field measurements can be made at whole dairies. These measurements and their accuracy are of interest for monitoring emissions of regulation compliance. Two studies have measured CH₄ emissions from a total of 18 whole dairies in the Netherlands and Canada (Tables 4 and A8). Measurement techniques included a mobile tunable diode laser sensor for a single drive-by measurement of each dairy (Hensen et al. 2005 and 2006) and an open path laser deployed for a few weeks at each dairy (McGinn and Beauchemin 2012). No studies measured N₂O from whole dairies.

When corrected for enteric emissions, mean CH₄ emissions from all other sources on the dairies were 309 ± 96 g hd⁻¹ d⁻¹, with a range of -209–1005 g hd⁻¹ d⁻¹ (Table 6). Winter emissions tended to be lower than summer emissions. The IPCC has no explicit Tier 2 approach for calculating emissions from entire dairies, but the equivalent is summing the predicted emissions from sources found on the dairies included in these studies. This study used default manure characteristic values for western Europe ($VS = 1861.5$ g hd⁻¹ y⁻¹, $B_o = 0.24$ m³ kg⁻¹) and cool MAT *MCFs* for corrals (1%), manure piles (0.5 %, value for static pile), slurry tanks (10 %), and barns (3 %, for deep bedding or pit storage) (IPCC 2006) to calculate a whole dairy emissions factor of 117 g hd⁻¹ d⁻¹. This factor is about 40 % of the field measurement mean (Table 6). The more general Tier 1 estimate of the CH₄ emissions rate from manure management, which

includes all sources except enteric emissions, is $\sim 57 \text{ g hd}^{-1} \text{ d}^{-1}$ for western Europe, or less than 20% of the measured value. The field measurement mean may be higher than the calculated annual average because the measurements were made mostly in early summer (Table A8) when emissions would be expected to be relatively high. Recognizing this mismatch, Hensen et al. (2005, 2006) calculated month-specific, rather than annual, CH_4 emissions rates using country-specific emissions factors and the Farm Greenhouse Gas model. Their calculated emissions rates were generally within 20–50 % of the measured emissions and were about evenly split between being too high and too low. Because the month-specific calculations addressed the seasonal bias that the Tier 2 approach could not, they were slightly more accurate.

For comparison, this study calculated a whole dairy value for California by summing the emissions from anaerobic lagoons, corrals, and solid manure piles, yielding a total value of $816 \text{ g hd}^{-1} \text{ d}^{-1}$ (Table 7). This value is comparable to that of the four dairies with the highest measured emissions (Table A8); 95% of these emissions came from anaerobic lagoons. This finding again highlights the large GHG emissions associated with anaerobic lagoons.

Implications for California Dairies

None of the studies included in the analysis above were done in California, and variables critical to development of representative emission factors, such as B_o and VS in equation 1, have not been determined for California dairies. Assuming model values accurately represent California dairies, GHG emissions from liquid manure management systems, particularly CH_4 from anaerobic lagoons, which were the highest total and per-cow source, represent the greatest opportunity for GHG emissions reduction from California dairies.

Before mitigation options and potential are discussed, several limitations of this comparison must be acknowledged. One weakness of the above analysis is the lack of emissions model parameters specific to California dairies. These parameters would reduce the error and uncertainty in estimates. Another weakness is the lack of field measurements to confirm emissions estimates. Model parameterization and estimate confirmation both require field measurements. Lack of field measurements in California is one of the greatest sources of uncertainty in the GHG inventory.

Gerber et al. (2013) report several options for CH_4 reduction from anaerobic lagoons and slurry systems:

- emplacement of a natural or induced crust,
- sealed containment with flaring of gas (simple anaerobic digestion),
- conversion to an advanced anaerobic digestion system with biogas collection for energy production, and
- reduction of the volume of manure in liquid storage systems while maintaining milk production.

The goal of the first three options is to convert CH_4 to climate-neutral CO_2 .

Crusts

Crusts may provide an oxidative environment in which methanotrophs consume CH_4 as it diffuses out of the liquid manure. These same conditions, however, may promote N_2O production. No studies have been made of the effect of crusts on GHG emissions from California liquid manure systems. Therefore, the efficacy of this approach is unknown and likely depends on climate and management practices.

Anaerobic Digestion

CARB (2011) GHG estimates indicate that the total CH₄ and N₂O emissions per head from anaerobic digestion is about 10% of the emissions from anaerobic lagoons (Table 3). Other benefits of anaerobic digesters include:

- production of energy for on-farm use (and possible purchase by utility companies) through the combustion of biogas (largely CH₄),
- decreased manure volume (which decreases subsequent handling costs and energy),
- improved fertilizer value of the effluent (discussed below),
- elimination of some pathogens (Kearney et al. 1993) and weed seeds (Johansen et al. 2013) from the manure, and
- decreased odors (USEPA 2011a; Massé et al. 2011; Camarillo et al. 2012).

The USEPA (2011b) estimated that adoption of anaerobic digesters by all the California dairies for which this technology is feasible (those with liquid manure management systems and > 500 hd, or approximately 900 farms with 1.35 million hd) could reduce California's CH₄ emissions by 7.7 Tg CO₂e y⁻¹, or 92% of the total CH₄ emissions from dairy manure management. In addition, these anaerobic digesters would be capable of producing more than 2 million MWh y⁻¹ (USEPA 2011b). According to the GHG equivalency calculator at www.epa.gov, this energy production offsets an additional 1.41 Tg CO₂e (considering CO₂ emissions from energy production).

Several obstacles have been encountered with implementation of anaerobic digesters, which has led to their abandonment and to reluctance of dairies to invest in them (CEC 2009; Camarillo et al. 2012). These obstacles include:

- issues with connectivity to utilities and reimbursement for excess energy production,
- long and sometimes complicated permitting processes,
- high construction and maintenance costs,
- issues related to design (leaking, corrosion, clogging, complexity),
- operation below maximum efficiency, and
- production of ozone precursors (particularly NO_x and VOCs) during the combustion of biogas collected from the digester.

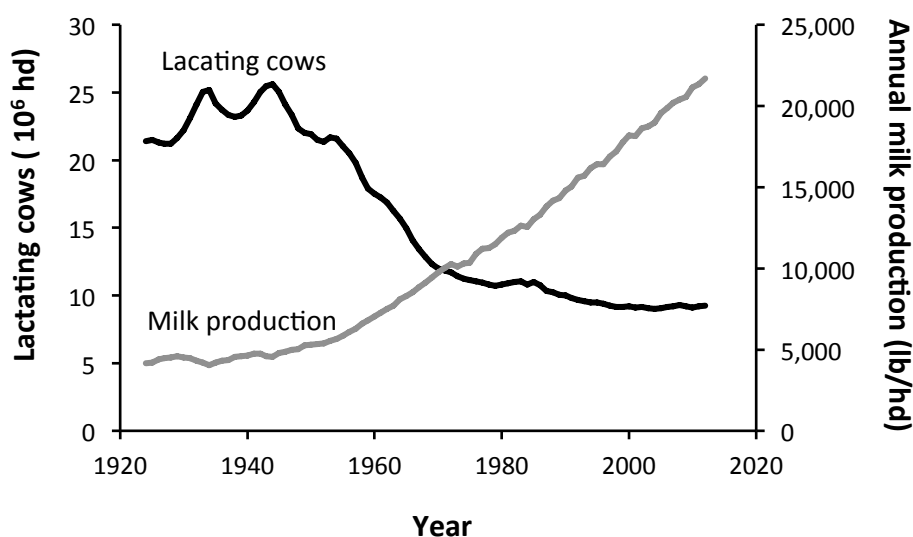
Although NO_x and VOC emissions are not a problem in all areas, they are regulated in many California counties and are costly to mitigate. Current work is focused on improving the economic viability of anaerobic digesters through legislation to streamline the permit process and facilitate cooperation of utilities to increase economic benefits. Other work is focused on improving anaerobic digester design for better performance and decreased emission of undesirable gases (Ward et al. 2008).

Decreased Volume of Manure in Liquid Systems

Decreasing the amount of manure in liquid management systems could be achieved by managing more manure in solid form or decreasing the total number of cows while retaining total milk production. Managing more manure in solid form would be feasible if dairies were still being constructed, but recent economic stresses have instead decreased the number of dairies (CDFA 2013). Thus, doing so would involve a major renovation of dairy infrastructure. A full life-cycle assessment (LCA, discussed below) would also be required to ensure that managing manure primarily as a solid would result in a net decrease in GHG emissions.

Decreasing the number of cows decreases both enteric emissions and manure production, but not necessarily at the expense of total milk production. Increasing milk production per cow increases the milk production per amount of GHG emissions from manure management and enteric fermentation. The number of cows in the United States has steadily declined as milk production has increased (Figure 7, <http://www.nass.usda.gov/>). Between 1944 and 2007, U.S. milk production efficiency increased such that, per billion kg of milk produced, the number of lactating cows decreased from 414,800 to 93,600, N excretion decreased from 17.47 Gg N to 7.91 Gg N, and CH₄ emissions decreased from 61.8 Gg CH₄ to 26.8 Gg CH₄ (Capper et al. 2009). Variation in the amount of milk produced per lactation and per cow over her lifetime remains large, indicating the high potential for genetic selection to further increase milk production efficiency while decreasing management costs and GHG emissions (Knapp et al. 2011).

Figure 4. Decreasing Number of Milk Cows and Increasing Production per Head in the United States between 1924 and 2013



Source: <http://www.nass.usda.gov/>.

Life-cycle Assessments of Dairy Manure Management

Quantifying the sinks and sources of C and N related to manure management on California dairies is critical for identifying management practices that reduce GHG emissions as well as for parameterizing LCAs. LCAs are powerful tools for guiding GHG reduction efforts in that they calculate the GHG emissions associated with all steps of production of a product, typically up to the point that the product leaves the farm (“cradle-to-gate”). They are especially important for determining the net effect of GHG management strategies, ensuring that they do not actually result in increased GHG emissions or emissions of other air pollutants. Parameterizing emissions models for California dairies is under way (Salas et al. 2008; CARB 2011), but constraining emissions factors and their relationship to manure management approaches will enable more confident, accurate estimates of emissions from dairies.

Numerous models have been developed for dairy and agricultural systems on the basis of varying combinations of emission factors, empirical relationships, and process-based calculations (see Table 1 in Li et al. 2012 for a summary of those developed for manure management). The utility of models is dependent on the availability of high-quality data for parameterization. Such data are lacking in most areas of California.

Manure-DNDC (Li et al. 2012) is a modification of the biogeochemical, process-based Denitrification-Decomposition (DNDC) model and links the core of DNDC to a virtual animal farm. Li et al. (2012) developed Manure-DNDC to predict GHG and NH₃ emissions from manure in storage and field application. The Canadian government has commissioned an integration of animal (dairy) emissions models and Manure-DNDC, which E. Kebreab and W. Salas (Applied GeoSolutions) are conducting. An integrated swine model for estimating air emissions from both animals and manure management is also in its final stages (E. Kebreab). Further work is necessary for integrating emissions models for dairy cow enteric emissions and manure management under conditions in California using local data wherever possible.

The DairyGHG model (Rotz et al. 2010) can perform partial LCA (cradle to farm gate) of dairy farms. It was used to compare the GHG costs of milk production for a hypothetical Pennsylvania dairy (Chianese et al. 2009a,b,c) as well as two hypothetical California dairies (Rotz et al. 2010). DairyGHG simulated significant differences in global warming potential for dairies utilizing different management practices. However, the lack of field data for comparison with, or calibration of, the California dairy simulations is problematic and indicates a significant knowledge gap.

The Integrated Farm System Model (ISFM; Rotz et al. 2011) was developed to track environmental impacts (P loss, NH₃ and hydrogen sulfide emissions) and economic costs for typical farming systems. It has since been expanded to account for GHG emissions. As with other models, calibration of the ISFM for California systems is limited, but this model is one of the few options for concurrently analyzing the economic and environmental impacts of management practices designed to decrease GHG emissions.

The selection of GHG mitigation practices requires consideration of the net effects of the practice on GHG and other criteria air pollutants. This analysis is best achieved through life-cycle assessments of various types of emissions, other environmental impacts, and economic impacts. Field and laboratory data necessary for constraining and confirming these assessments are nearly absent for California. There is no question that decreasing emissions from anaerobic lagoons will have a significant impact on total GHG emissions, but the absolute decrease in emissions is uncertain without well-constrained models to determine it.

LAND APPLICATION OF MANURE

Dairy manure that does not volatilize, decompose, or evaporate during on-dairy management is eventually applied to agricultural land as a fertilizer, and liquid manure can be used for irrigation. Agricultural soils typically emit little CH₄ with the exception of rice cultivation. This report focuses on the sequestration of C in the soil, which decreases the emission of CO₂ and CH₄ associated with manure decomposition. Nitrous oxide emissions from land application of manure are discussed by Culman et al. (2014).

Using manure as fertilizer rather than inorganic fertilizers has benefits and challenges. Benefits include decreasing or eliminating entirely the need for inorganic fertilizer and increasing soil C. Compared with manure, synthetic fertilizers have a higher energy cost and global warming potential associated with their production and application, assuming similar transport distances (Wiens et al. 2008; Eagle and Olander 2012).

Long-term intensive cultivation and inorganic fertilization of agricultural fields has significantly reduced soil organic C content by more than 1000 Tg in the United States (Kern and Johnson 1993). This soil C was largely released to the atmosphere as CO₂. Because soil C was not replenished, agricultural soils are a net source of greenhouse gases. As described below, manure additions can increase soil C storage through

direct (manure) and indirect (plant growth) inputs (Eagle and Olander 2012). Increased soil organic matter decreases bulk density, increases water and nutrient holding capacity, and improves aggregation, all of which can promote plant growth (Brady and Weil 2010). Most important to this report, soil organic matter also sequesters C from the atmosphere.

One of the greatest challenges of fertilization with manure is that a large portion of the N in manure is in the organic form and is released more slowly and less predictably to the soil through decomposition than through inorganic fertilization (Van Kessel and Reeves 2002). Because of this slow release, farmers are less able to match N availability with crop demand, which is typically non-linear through the growing season (Gastal and Lemaire 2002). Manure is also a composite fertilizer containing multiple nutrients that are added together. Thus, the optimum application rate for N may under- or overapply P and K, leading to nutrient deficiencies or pollution (Van Kessel and Reeves 2002). Lastly, due to its solids content, liquid manure may require application techniques different from those for inorganic fertilizer dissolved in irrigation water (Pfoest et al. 2001).

Carbon Storage in Agricultural Fields through Manure Addition

Manure addition has been shown to increase soil C storage in California (Poudel et al. 2001; Kong et al. 2005; Kong et al. 2009) and in fields in other regions with a Mediterranean climate (Aguilera et al. 2013a). In 10 different cropping systems after 10 years of management at the Long-Term Research of Agricultural Systems (LTRAS) site, organic management (legume cover cropping and composted manure amendment) resulted in an average SOC sequestration rate of $0.56 \text{ Mg C ha}^{-1} \text{ y}^{-1}$ in the top 15 cm of the soil, whereas most other management systems were small sources or significantly smaller sinks (less than $\pm 0.06 \text{ Mg C ha}^{-1} \text{ y}^{-1}$) (Kong et al. 2005). Much of the soil C gain in the organic fields was due to high C inputs ($89.6 \text{ Mg C ha}^{-1}$ over the 10 years). Soil C accumulation increased the number and stability of macroaggregates in soils, which may lead to longer C residence times in soils. A comparison between fields revealed the importance of amendment quality in soil C storage capacity. It also revealed that fields with manure amendment were more efficient at storing C than similarly managed fields without manure amendment (Kong et al. 2005).

A similar experiment at the Sustainable Agricultural Farming Systems (SAFS) research site near Davis, California, showed that fields organically farmed for 10 years using poultry manure increased soil C concentrations in the top 15 cm of soil compared to conventional fields (12.6 g kg^{-1} vs. 9.6 g kg^{-1} , respectively) (Poudel et al. 2001). Assuming a soil bulk density of 1.2 g cm^{-3} (the average for the nearby LTRAS site; Burger et al. 2005), the difference in C content between synthetic fertilization and manure amendment was 5.4 Mg C ha^{-1} , equivalent to $0.54 \text{ Mg C ha}^{-1} \text{ y}^{-1}$ or $2 \text{ Mg CO}_2\text{e ha}^{-1} \text{ y}^{-1}$. Soil C can increase rapidly; at the Russell Ranch experimental site, also near Davis, organic management (composted dairy manure amendment) increased SOC by approximately 5 Mg C ha^{-1} compared with conventional and low-input management one year after conversion, regardless of tillage (Kong et al. 2009).

The sequestration rates reviewed above are within the range found in other Mediterranean agricultural systems, where C sequestration rates ranged from 0.5 to $2.3 \text{ Mg C ha}^{-1} \text{ y}^{-1}$ following manure addition (Aguilera et al. 2013a). Carbon sequestration in fields with livestock slurry amendment, however, showed no significant difference from conventional fields. Though this finding was based on only three studies, it is consistent with results from other regions. Slurry typically has relatively high mineralized N content and low C content which promotes microbial growth and the decomposition of soil organic matter and release, rather than storage, of N_2O (Aguilera et al. 2013a).

The California-based field studies described above provide a useful starting point for understanding C sequestration in agricultural fields amended with manure, but they represent a small cross-section of

typical agricultural practices in the state for a limited geographic area on only three different soil series. Carefully controlled and replicated field studies in a wider range of the state's bioclimatic zones and on more soil types are needed to expand this analysis.

Impacts of Manure Type

Manure composition and form can have significant effects on N₂O production and C storage, particularly in Mediterranean climates where soil processes are sensitive to changes in soil moisture associated with slurry application (Aguilera et al. 2013a,b). Anaerobic digester output has received less attention than solid and liquid manures, but its availability in California could increase. As of 2009, at least 10 California dairies had anaerobic digesters as a result of the Dairy Power Production Program undertaken by the California Energy Commission (CEC) as part of Senate Bill 5X (CEC 2009). In September 2012, California passed California Senate Bill 1122 (SB1122), which directs the California Public Utilities Commission to require the utility industry to produce at least 250 megawatts from bioenergy operations that commence on or after June 1, 2013. Much of this mandate is expected to be met with biogas production from anaerobic digesters utilizing municipal waste, food processing waste, and livestock manure.

Few studies have compared the GHG emissions from land application of anaerobic digestate and other manure types. Total GHG emissions (CH₄ and N₂O) from pilot-scale manure storage and land application in Austria were lower for anaerobically digested slurry than for untreated slurry, separated slurry, slurry covered with straw while in a tank, and aerated slurry (Amon et al. 2006). Similar findings were found on a xeric, irrigated agricultural field in eastern Washington (Collins et al. 2011), where cumulative N₂O emissions were greater for dairy slurry- and digested effluent-amended fields than for fields treated with synthetic fertilizer. The total global warming potential, however, was lowest for those amended with the digested effluent (Table 10) and was approximately 20 kg CO₂e ha⁻¹ lower than for fields amended with synthetic fertilizer, and 25–130 kg CO₂e ha⁻¹ lower than for fields amended with untreated liquid manure.

Table 10. Net Global Warming Potential (kg CO₂e ha⁻¹, Mean ± Standard Error, Considering N₂O and CH₄) of Corn Fields near Prosser, Washington, with Different Amendments Over Two Study Seasons

Treatment	2007	2008
Fallow	63.3 ± 17	38.0 ± 10
Unfertilized	51.4 ± 22	86.9 ± 15
Synthetic fertilizer	114.0 ± 12	225.1 ± 40
Digested fiber	115.0 ± 35	62.5 ± 17
Digested effluent	90.6 ± 22	192.5 ± 69
Liquid manure	219 ± 18	217.7 ± 22

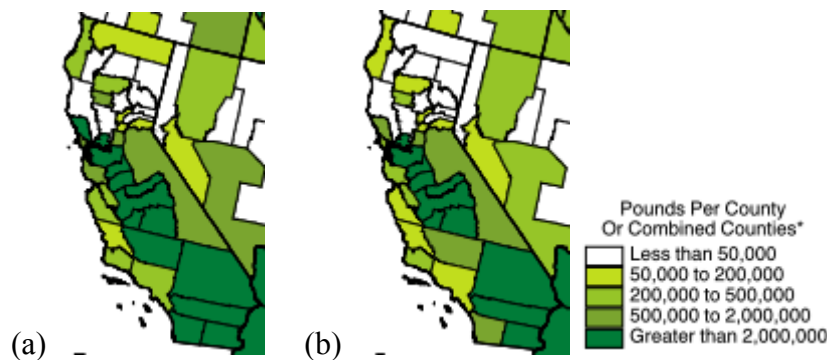
Source: Table 3 in Collins et al. (2011).

The consistency of results in the Amon et al. (2006) and Collins et al. (2011) studies suggests that land application of the outputs from anaerobic digestion may lower GHG emissions and maintain or improve crop yields under conditions and practices common in California. Very little field work has been done in California to quantify GHG emissions related to land application of manure and the products of anaerobic digestion. Using rates estimated outside California, application of anaerobic digestate, rather than synthetic fertilizer or liquid manure, could decrease GHG emissions by 20-130 kg CO₂e ha⁻¹ y⁻¹.

Limitations to Land Application of Manure

Over the last few decades, the increasing consolidation of the California dairy industry has resulted in more large CAFOs and their concentration in the San Joaquin Valley and other areas (Kellogg et al. 2000; CDFA 2013). This concentration may create a water pollution issue as more manure is produced in an area with insufficient land for application at levels that do not compromise water quality. The amount of N and P in livestock manure by farm and by county in the United States has been estimated and compared with the estimated land available (permanent pasture and agricultural land associated with 24 crops) to receive that manure (Kellogg et al. 2000). The southern valleys of California stand out as regions of high excess N and P, assuming no manure export from farms (maps 28 and 30, Kellogg et al. 2000, reproduced in Figure 5a,b).

Figure 5. Excess Manure N (a) and P (b) by County in 1997 in California, Assuming No Manure Export from Farms

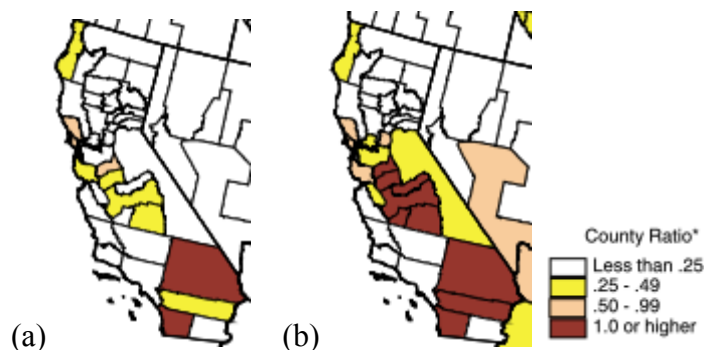


Source: Maps 28 and 30, respectively, in Kellogg et al. (2000).

Note: Includes all livestock.

Considered by county and assuming a more realistic scenario in which livestock operations could dispose of excess manure on other land within the county, the ratios of nutrient production to nutrient assimilation capacity for both N and P were less than 1 (where > 1 indicates excess nutrients and < 1 indicates crop assimilation of all the manure nutrients) for almost all California counties in 1997. San Bernardino and San Diego counties had the potential for excess N (Map 35 in Kellogg et al. 2000, reproduced in Figure 6a); several more counties had the potential for excess P (Map 37 in Kellogg et al. 2000, reproduced in Figure 6b). Total farm-level excess N was estimated as $\sim 61,000$ Mg in California in 1997 (Kellogg et al. 2000). Excess N addition to soils increases N_2O emissions.

Figure 6. Ratio of Manure N Produced to Assimilation Capacity (a) and Ratio of Manure P Produced to Assimilation Capacity (b) in 1997 in California



Source: Maps 35 and 36, respectively, in Kellogg et al. (2000).

The above analysis may not accurately reflect the current extent of nutrient excess in California due to the more recent implementation of the NRCS CNMP program, which limits manure N application to 140% of crop N needs. This reduction is significant and necessitates more off-farm options for manure disposal (Owen and Silver, in prep). Additionally, other constituents of some manure (e.g., hormones, antibiotics, pathogens, salts) have the potential to affect soil and water systems.

Application of Manure to Rangeland

Though most livestock manure is eventually applied to agricultural fields, some is applied to rangelands. This practice may become more common as both a means of manure disposal and a global warming mitigation strategy. Manure application rates to agricultural fields on dairies are now limited by estimates of plant nutrient demand. Thus, application to rangelands may be a viable alternative and may offer the co-benefits of increasing forage production and soil carbon sequestration (see DeLonge, Owen, and Silver 2014).

Modeling Land Application of Manure

Biogeochemical models can be used to estimate the net effect of different management practices on GHG emissions and soil C and N, provided they are adequately parameterized for the soil and crop types of interest. This study focuses on models of soil C. De Gryze et al. (2010) calibrated the DAYCENT model with data from four long-term agricultural experiments. According to their estimates, soil C sequestration is much greater in organic fields than in conventional fields, regardless of tillage (Table 11). These estimates suggest that switching from conventional management to manure application could decrease total GHG emissions by $> 4 \text{ Mg CO}_2\text{e ha}^{-1} \text{ y}^{-1}$ when soil C is considered. Estimates from DAYCENT generally agreed with field measurements, with variability introduced by inter-annual climate variability and by factors related to crop growth that are not included in the model. They found that manure amendments had lower potential GHG emissions than inorganic fertilizers.

Table 11. DAYCENT-Derived Estimates for Change in Soil Organic C in the Top 1.5 m of Soil (Δ SOC) and Net Soil GHG Flux for the Long-Term Research on Agricultural Systems Study Site

Management	Δ SOC (kg C ha ⁻¹ y ⁻¹)	Net greenhouse gas flux (kg CO ₂ e ha ⁻¹ y ⁻¹)
<i>Standard tillage</i>		
Synthetic	95 ± 46	1081 ± 192
Low input	315 ± 46	9 ± 192
Organic	1324 ± 46	-3496 ± 192
<i>Conservation tillage</i>		
Synthetic	47 ± 87	1182 ± 391
Low input	321 ± 87	-192 ± 391
Organic	1279 ± 87	-3349 ± 391

Source: Table 5 in De Gryze et al. (2010).

Note: Positive values of Δ SOC indicate an increase of C in the soil, whereas positive values for net soil GHG flux indicate loss to the atmosphere. Synthetic = synthetic fertilizer; low input = less synthetic fertilizer and cover cropping; organic = poultry manure fertilizer and cover cropping.

Alternative management practices were modeled for eight crops grown in ten counties in the Central Valley of California to determine their effect on crop yields and GHG emissions (De Gryze et al. 2011). Using manure instead of synthetic fertilizer decreased yields by less than 4%, in most cases, while decreasing GHG emissions by 1.16 Mg CO₂e ha⁻¹ y⁻¹ in the Sacramento Valley and 0.5 Mg CO₂e ha⁻¹ y⁻¹ in the San Joaquin Valley. A large portion of the decrease in GHG emissions with the switch from synthetic fertilizer to manure was due to accumulation of soil organic C, which could be reversed with increased tillage. These estimated changes in emissions did not include changes associated with decreased synthetic fertilizer production or changes in fuel use with adoption of the alternative management practices. Detailed manure characteristics used in the model were not specified; results from LTRAS were used to constrain the model (De Gryze et al. 2010), but this study used composted poultry manure, which may not be applicable to dairy manure-amended fields.

In summary, modeling tools that can estimate GHG emissions from land application of manure have been developed and are being refined and parameterized for California. Modeling results suggest that using manure rather than synthetic fertilizer could result in a decrease of 0.5-4 Mg CO₂e ha⁻¹ y⁻¹. Complete life-cycle assessment is needed to determine the total GHG reduction associated with this switch. Such assessment would address fuel use for land management and manure transportation, synthetic fertilizer production, and the emissions of other pollutants of concern such as NH₃. Model application to California is limited by the lack of field data to constrain the emissions factors and process models at their core. California's diverse agriculture and soils, Mediterranean climate, and extensive use of anaerobic lagoons and flood irrigation are substantial challenges that the models must overcome.

Implications for California Agriculture

The results discussed above suggest that GHG emissions from land application of solid manure or liquid effluent from anaerobic digesters are lower than from fields treated with synthetic fertilizers. Modeling results suggest that using manure rather than synthetic fertilizer could result in a decrease of 0.5-4 Mg CO₂e ha⁻¹ y⁻¹. Unfortunately, none of the studies measured or modeled GHG emissions from the most common land application practice on California dairies: furrow irrigation with diluted lagoon effluent. This practice may increase emissions of both CH₄ and N₂O.

Field studies of California's typical dairy agricultural fields are needed to improve model estimates and identify practices that decrease GHG emissions. Particularly critical are long-term agricultural experiments that are representative of practices on dairy-associated fields, including furrow irrigation with diluted liquid manure from anaerobic lagoons.

KEY FINDINGS AND RESEARCH PRIORITIES

The literature review described above identified significant opportunities for GHG mitigation:

- **Livestock manure, particularly dairy manure stored in anaerobic lagoons, is likely to be the largest source of GHG emissions from California agriculture.** Manure emissions potentially could be decreased more than 90% if lagoons are capped (and their methane emissions flared), farms transition to anaerobic digesters, or both.
- **Decreasing the amount of manure treated in liquid systems, whether by changing manure management practices or decreasing the number of cows, would also decrease total GHG emissions.** Anaerobic lagoons, the most common on-farm dairy manure storage practice in California, have more than 10 times the global warming potential of solid manure piles. Milk production has been steadily increasing despite a national decrease in cow population. Future advances in breeding and feed management could further reduce the amount of manure produced for the same or increased milk production.
- **Land amended with manure had lower emissions than fields managed with synthetic fertilizer and had consistently greater soil C content. Compared with other manure management practices, land application decreased GHG emissions associated with manure decomposition.** Research from outside California suggests that anaerobic lagoon effluent may result in greater emissions following field application compared with solid manure and anaerobic digester effluent, yet both types of effluent have lower emissions than fields managed with synthetic fertilizer.

We identified several key research needs:

- **California-specific measurements of GHG fluxes from manure management are needed, particularly from anaerobic lagoons on dairies, anaerobic digesters, fields with furrow irrigation and fertilization, and land-applied manure and anaerobic digester effluent.** No GHG emissions measurements from manure management or land application of dairy manure in California have been published. These data are needed to improve GHG emissions estimates and to identify best management practices. Long-term measurements of soil C and N content, CH₄ and N₂O emissions, and crop yields (including pasture productivity) in conjunction with local climate data, management data, and soil characteristics are especially critical.
- **Comparisons of field measurement techniques are needed to facilitate comparisons across studies and scales of analysis.** This literature review revealed wide variations in GHG emissions measurements from some sources—variations that may be the result of technique, rather than differences in actual emissions. Scientists must critically evaluate methodologies to identify the measurement techniques that most accurately quantify GHG emissions from heterogeneous manure storage systems and from areas where manure has been applied.
- **Process-based models are increasingly important in informing policy decisions, but must be parameterized for the range of manure management practices, crops, and irrigation systems found in California.** The lack of field and laboratory research on GHG sources and sinks, and the mechanisms controlling these dynamics in California severely limit the utility of models. Integration of animal, soil, and, ideally, crop biogeochemical models is needed at local and regional scales.
- **Surveys of manure management practices are needed for all major dairying regions of California.** These surveys would expand the work of Meyer et al. (2011). Accurate emissions estimates require better estimates of (1) the amount of manure C and N under the various

management practices, (2) the spatial distribution of these practices across California, and (3) the length of storage and ultimate fate of manure. These inputs are critical for identifying trends in manure management in the dairy industry as well as for quantifying California's excess manure nutrient production, which may affect land application practices and have economic impacts on farmers.

The California dairy industry is already at the forefront of environmental sustainability innovations, and its successful mitigation of GHG emissions, coupled with economic growth, can serve as an example to other dairy industries across the world. More emissions data and improved models, however, are needed to accurately quantify this achievement and optimize energy, crop, and economic returns from manure management.

APPENDIX A: DATA FOR DAIRY EMISSIONS STUDIES

Table A1. Study location, measurement technique, and climate data for the 32 studies of dairy emissions.

APPENDIX B: TERMINOLOGY

Anaerobic lagoon: Large, earthen ponds that hold the liquid fraction after mechanical or gravity-driven separation of the manure plus wash water.

Anaerobic digester: A system in which manure, wash water, and often other waste materials, anaerobically decompose, resulting in methane (biogas) production. This methane is collected and either flared (combusted so that it is released as climate-neutral carbon dioxide) or used to power a generator for on-farm energy (electricity, heat, or both) production. Designs range from covers for existing anaerobic lagoons to temperature-controlled batch reactors (IPCC 2006).

Barn floors: Heterogeneous surfaces in barns, including paved or slatted-floor areas for livestock movement, farmer access, and manure management, as well as stalls with some sort of soft bedding where livestock can rest.

Confined animal feeding operation (CAFO): Livestock farms that raise a large number of animals in corrals, pens, barns, or a combination of these structures.

Corral: A dirt-floored dry lot.

Daily spread: Manure is collected daily from the confinement facility and applied to pasture or cropland (IPCC 2006).

Dairy cow: A cow that has calved and is lactating or is briefly between lactations.

Dairy heifer: A cow that has not yet calved.

Deep pit: Manure is captured from a confinement facility, usually through a slatted floor, and stored in a tank with minimal water addition (IPCC 2006).

Dry lot: An open, paved or unpaved confinement area with few plants from which manure is scraped and removed periodically (IPCC 2006).

Enteric fermentation: Microbially mediated, anaerobic digestion of plant material in the rumen of ruminant livestock.

Hardstandings: Areas with solid surfaces, such as concrete, used as corrals or as temporary holding pens.

Liquid manure: Manure with less than 20% solid content (IPCC 2006).

Loafing pens: Corrals on dairies in which milk cows rest and which are typically dirt-floored and spread with some sort of bedding material, often dried manure solids.

Manure management: Activities related to collecting and storing manure on livestock facilities.

Slurry tanks/settling pond: Storage containers filled with unseparated, minimally diluted livestock manure with a higher solids content than anaerobic lagoons.

Soil management: Activities related to fertilization, cultivation, and organic matter addition of agricultural soils.

Solid manure: Manure with greater than 20% solid content (IPCC 2006).

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