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SPECIAL TOPICS—Mitigation of methane and nitrous oxide emissions from animal operations: II. A review of manure management mitigation options[1]

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ABSTRACT: This review analyzes published data on manure management practices used to mitigate methane (CH_4) and nitrous oxide (N_2O) emissions from animal operations. Reducing excreted nitrogen (N) and degradable organic carbon (C) by diet manipulation to improve the balance of nutrient inputs with production is an effective practice to reduce CH_4 and N_2O emissions. Most CH_4 is produced during manure storage; therefore, reducing storage time, lowering manure temperature by storing it outside during colder seasons, and capturing and combusting the CH_4 produced during storage are effective practices to reduce CH_4 emission. Anaerobic digestion with combustion of the gas produced is effective in reducing CH_4 emission and organic C content of manure; this increases readily available C and N for microbial processes creating little CH_4 and increased N_2O emissions following land application. Nitrous oxide emission occurs following land application as a byproduct of nitrification and dentrification processes in the soil, but these processes may also occur in compost, biofilter materials, and permeable storage covers. These microbial processes depend on temperature, moisture content, availability of easily degradable organic C, and oxidation status of the environment, which make N_2O emissions and mitigation results highly variable. Managing the fate of ammoniacal N is essential to the success of N_2O and CH_4 mitigation because ammonia is an important component in the cycling of N through manure, soil, crops, and animal feeds. Manure application techniques such as subsurface injection reduce ammonia and CH_4 emissions but can result in increased N_2O emissions. Injection works well when combined with anaerobic digestion and solids separation by improving infiltration. Additives such as urease and nitrification inhibitors that inhibit microbial processes have mixed results but are generally effective in controlling N_2O emission from intensive grazing systems. Matching plant nutrient requirements with manure fertilization, managing grazing intensity, and using cover crops are effective practices to increase plant N uptake and reduce N_2O emissions. Due to system interactions, mitigation practices that reduce emissions in one stage of the manure management process may increase emissions elsewhere, so mitigation practices must be evaluated at the whole farm level.

Key words: manure, methane, mitigation, nitrous oxide

INTRODUCTION

Climate change associated with emissions of greenhouse gases (**GHG**) resulting from human activities is noted as the defining human development issue currently (UNDP, 2008). Through the production of feed, growth of productive animals and the supporting herd, and the disposal of animal waste, livestock production contributes 8 to 18% (depending on the estimation method used) of global GHG emissions (O'Mara, 2011)

and emissions from animal agriculture are expected to increase 17% by the year 2020 (USEPA, 2006). Significant reduction of GHG emissions can be achieved through intensification of animal production and improved production technologies in less developed countries and the use of novel mitigation practices in modern production systems (Smith et al., 2007; O'Mara, 2011).

This article reviews the direct effect of animal manure mitigation practices on emissions of the major non–carbon dioxide (**non-CO$_2$**) GHG, methane (**CH$_4$**) and nitrous oxide (**N$_2$O**), focusing on experimental and field evaluation data and excluding for the most part modeling and simulation studies. The goal was to gather experimental information to support the development of whole farm analyses and life cycle assessments (**LCA**) necessary to identify opportunities for synergies and consequences of interactions derived from the use of these practices. Interactions among mitigation practices for individual components of livestock production systems are discussed in Gerber et al. (2013).

LIVESTOCK MANURE AND EMISSIONS

Animal manure is a nutrient resource containing most of the essential elements required for plant growth and can be a significant source of N in both intensive and subsistence production systems. Application of manure to cropland increases soil OM, microbial biomass, and mineralization rate (Spiehs et al., 2010; Langmeier et al., 2002) and improves a number of soil properties including soil tilth, water-holding capacity, oxygen content, and fertility; it also reduces soil erosion, restores eroded croplands, reduces nutrient leaching, and increases crop yields (Khaleel et al., 1981; Araji et al., 2001). Animal manure is an alternative to energy intensive and high-cost synthetic fertilizer and can be a very effective fertilizer source when the available nutrient content and mineralization rate are synchronized with crop nutrient uptake.

Unmanaged accumulation of organic wastes, however, presents environmental and health concerns for humans and animals. Concerns include leaching of nitrate (**NO$_3$$^-$**) and pathogens to ground water, unbalanced algal growth and eutrophication of surface water and deterioration of sensitive ecosystems, degradation of soil production potential through accumulation of nutrients, salts, and metals, and emission of gases considered a health and environmental risk (Gerber et al., 2005; Steinfeld et al., 2006; USEPA, 2011).

Manure management refers to all activities, decisions, and components used to handle, store, and dispose of feces and urine from livestock with the goal of preserving and recycling the nutrients in the livestock production system (Brandjes et al., 1996; IPCC, 2006a). This includes manure accumulation and collection in buildings, storage, processing, and application to cropland as well as deposition in pastures and rangelands in grazing systems.

Methane and Nitrous Oxide Emissions. In ruminant production systems, enteric CH$_4$ production is the largest contributor to GHG emissions followed by CH$_4$ from manure and in beef feedlot systems, N$_2$O from pen surface, and N$_2$O emissions from soils. Emissions from nonruminant livestock systems are less than that of ruminants and are mostly CH$_4$ and N$_2$O from manure storage and land application (Hristov et al., 2013b).

The contribution of manure management to global GHG emission was estimated by Steinfeld et al. (2006) to be 2.2 Gt of CO$_2$ warming potential equivalents (**CO$_2$e**) per year. Methane emissions from manure storage were estimated to be 470 Mt CO$_2$e/yr in 2010 with an expected 11% increase by 2020 and N$_2$O emissions from fertilizer use, manure application, and deposition by grazing livestock were estimated at 2,482 Mt CO$_2$e/yr in 2010 with an expected 18% increase by 2020 (USEPA, 2006). Nitrous oxide emissions from soil application of manure are a major contributor to total GHG emissions from agriculture (Davidson, 2009) with animal waste representing 30 to 50% of the global agricultural N$_2$O emissions (Oenema et al., 2005). Both CH$_4$ and N$_2$O are powerful GHG with global warming potentials (**GWP**) of 25 and 298 kg CO$_2$e/kg, respectively (Solomon et al., 2007).

Most of the CH$_4$ emission from manure is produced under anaerobic conditions during storage with very little following land application. Manure produces less CH$_4$ when handled as a solid (e.g., in stacks or pits) or when deposited on pasture or rangelands (USEPA, 2005). Therefore, opportunities to reduce CH$_4$ emission are centered on preventing anaerobic conditions during storage or capturing and transforming the CH$_4$ that is produced, if anaerobic conditions are present. Data summarized by Chianese et al. (2009) indicate average CH$_4$ emissions from covered slurry, uncovered slurry, and stacked manure to be 6.5, 5.4, and 2.3 kg/m^2 per yr although rates vary with temperature and time in storage.

Direct emissions of N$_2$O from manure storage are small when compared with CH$_4$ emissions. For N$_2$O emissions to occur, manure must first be handled aerobically where ammonium (**NH$_4$$^+$**) or organic N is converted to NO$_3$$^-$ and nitrite (**NO$_2$$^-$**) during nitrification and then handled anaerobically where the NO$_3$$^-$ and NO$_2$$^-$ are reduced to elemental N (**N$_2$**), with intermediate production of N$_2$O and nitric oxide (**NO**) through denitrification (USEPA, 2010). Most of the N$_2$O resulting from manure is produced in manure-amended soils through microbial nitrification under aerobic conditions and partial denitrification under anaerobic conditions, with denitrification generally producing the larger quantity of N$_2$O (Tisdale et al., 1993; USEPA, 2010).

Losses of N_2O from the pen surface of open-lot dairy or beef feedlot facilities, however, can be significant. The fact that a large amount, up to 50%, of the excreted N by beef cattle is not recovered in manure has been well documented for various geographic locations (Loh et al., 2008; Cole and Todd, 2009). Most of these losses are as ammonia (**NH_3**), but N_2O emissions are also significant (Leytem et al., 2011; Rahman et al., 2013) and depend on a variety of factors, including surface conditions (Aguilar et al., 2011). Manure management practices in beef feedlots vary, but usually pens are cleaned when animals are marketed (i.e., several times a year) or once a year (Eghball and Power, 1994), which creates conditions for NH_3 and GHG emissions off the pen surface. Recently, Rahman et al. (2013) reported CH_4, CO_2, and N_2O emission rates from the pen surface of a North Dakota beef feedlot of 38 g, 17 kg, and 26 g/head per d, respectively. Somewhat lower N_2O emissions were reported for an open-lot dairy in southern Idaho: 0.13, 0.49, 28.1, and 0.01 kg/cow per d (NH_3, CH_4, CO_2, and N_2O, respectively; Leytem et al., 2011). Considering the much greater GWP of N_2O, compared with CH_4 (Solomon et al., 2007), the emissions of these 2 GHG were comparable in that and other studies (Boadi et al., 2004). Comparatively lower N_2O emissions from a free-stall dairy in the Texas panhandle were reported by Borhan et al. (2011): 836, 5,573, and 3.4 g/head per d (CH_4, CO_2, and N_2O, respectively). In the same study, emissions from a beef feedlot were reported as 3.8, 1,399, and 0.68 g/head per d, respectively. The pen surface was estimated to contribute about 84% of the aggregate N_2O emission in this study.

Manure contains most elements necessary for stimulating soil nitrification and denitrification processes that form N_2O. These processes are transient, depending on the amount and form of available N (NH_4^+ or NO_3^-), soil oxidation–reduction potential, degradable C sources, soil temperature, water content, and microbial population (Cavigelli and Parkin, 2012). Denitrifying organisms can further reduce N_2O to N_2 at rates dependent on soil conditions, with multiple factors controlling the ratio of N_2O to N_2 produced. The fraction of N completely reduced to N_2 also increases as soil water content approaches saturation. Nitrous oxide can also be produced indirectly when manure N is lost through volatilization as NH_3, NO, and nitrogen dioxide (**NO_2**) and is nitrified and denitrified in soil following redeposition (USEPA, 2010).

Being a result of microbial processes, the emission of N_2O is highly variable as influenced by environmental and metabolic factors, which makes measurement of mitigation effects difficult. Nonetheless, results of adopting mitigation practices can be estimated using the potential N_2O emission reductions obtained when optimal conditions for nitrification and denitrification are assumed. This approach makes it possible to gauge the effect of mitigation practices and their interactions within the livestock production system.

Due to the nature of the antagonistic processes resulting in CH_4 and N_2O emissions (CH_4 is produced under anaerobic conditions whereas production of N_2O requires sufficient levels of oxygen), some practices that result in the reduction of CH_4 production increase N_2O emissions. An example is the aeration of manure during storage to reduce CH_4 emissions. This process may increase N_2O emissions when aeration rate is sufficient to create an aerobic environment.

Opportunities to reduce N_2O and CH_4 emissions from livestock manure are illustrated in Fig. 1 and 2 according to the process of formation and emission of CH_4 and N_2O and the flow of organic C and N during the animal production cycle. The review of manure CH_4 and N_2O mitigation practices based on experimental and commercial scale applications is summarized in Table 1. Data related to the mitigation practices presented in this review may not account for losses, conversions or other differences caused by preceding processes in the system, study duration, and climatic impacts such as winter and summer differences.

There is still much to learn about the benefits of particular mitigation practices, the effect of combining mitigation practices, the response of environmental indicators such as nutrient conversion, volatilization, leaching, erosion, etc., and the effect on the environmental and financial performance of the production system (farm) as a whole. It is common for one environmental benefit of a technology to negatively impact processes in another area (Gerber et al., 2013).

Ammonia: An Important Component in Greenhouse Gas Mitigation. Although not a GHG, NH_3 (and its ionized form, NH_4^+) is an important component of the manure N cycle. Ammonium (a large fraction of the N in manure) is the first product of decomposition of urea through the action of the microbial enzyme urease after urine is deposited on barn floors and pastures. Urease is abundant in fecal matter and in soil, and thus urea excreted in urine is rapidly converted to NH_4^+ when the environmental conditions (temperature, pH) are favorable. Ammonium N can be converted under aerobic conditions to NO_3^- and both forms of N are readily available to plants whereas organic forms of manure N are generally not readily available (Beegle et al., 2008).

Ammonium N is also the carrier of rapidly available N in the soil and a necessary precursor in the process that leads to N_2O emissions from application of manure and fertilizers and urine deposition in pastures (Tisdale et al., 1993; de Klein et al., 2001). Ammonia is a volatile gas that escapes to the atmosphere reducing the amount of N transported to the soil, which may offset the benefits of manure storage and land application mitigation practices. Simultaneously, NH_3 that escapes to the at-

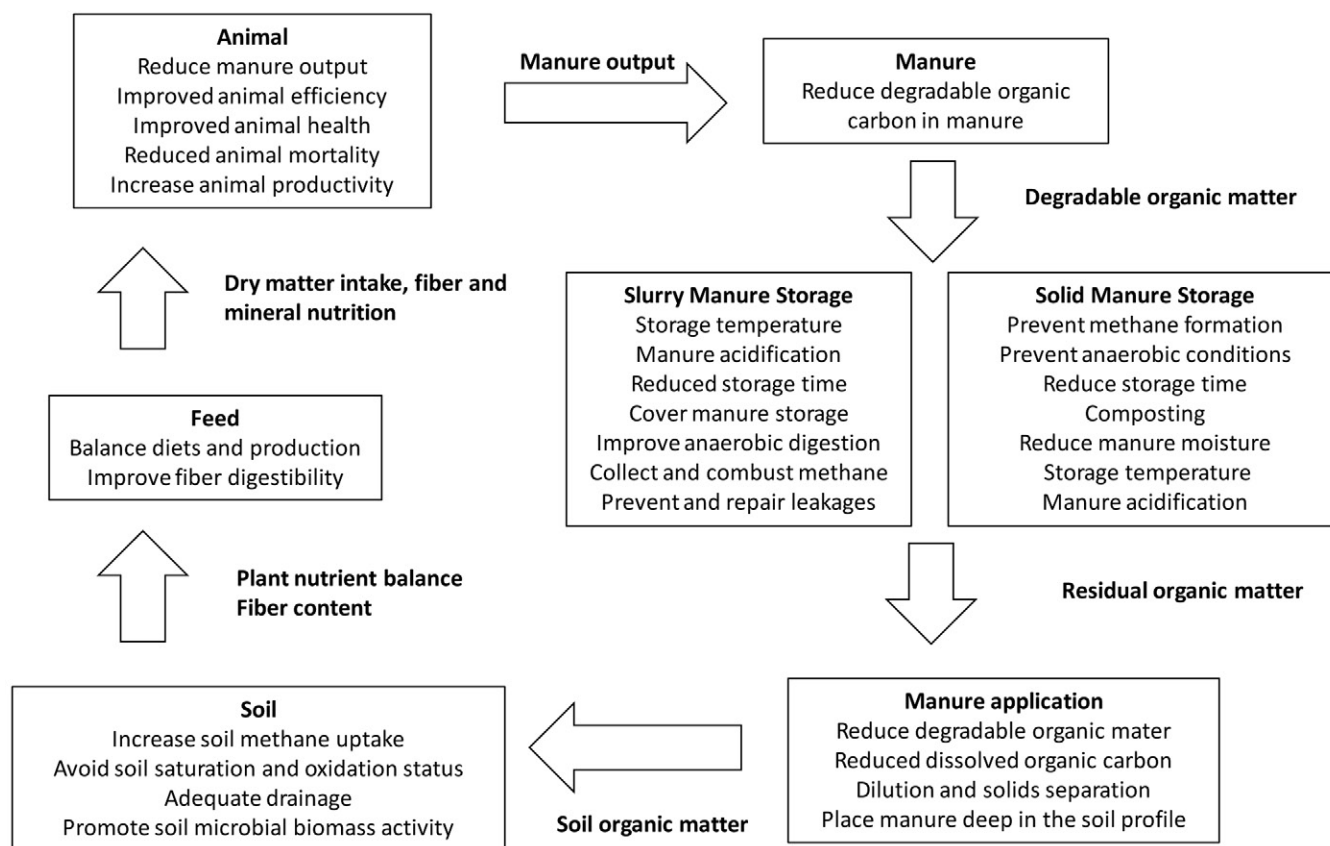## Opportunities to mitigate methane from livestock manure

Figure 1. The flow of organic carbon through the livestock production system including opportunities to mitigate methane from livestock manure.

mosphere can form particulate matter that may return to soil through dry or wet deposition. This deposition can harm sensitive ecosystems and contribute to N runoff and groundwater pollution and be converted into N_2O through denitrification (Galloway et al., 2004; USEPA, 2010; Hristov et al., 2013b).

Ammonia volatilization is generally the largest pathway of loss for manure N (Harper et al., 2004; Lee et al., 2011a), with losses typically accounting for 30 to 70% of the NH_4^+ content of cattle manure (Thompson and Meisinger, 2002). Nitrogen emissions can also be in the form of N_2 (Harper et al., 2004) but losses as N_2 have not been well quantified. Thompson et al. (1987) estimated that 2 to 12% of applied manure in English grassland was lost to denitrification ($N_2 + N_2O$) with surface application and 7 to 21% was lost with injection of manure. The portion of manure N lost as N_2O is relatively low, generally below 2 to 3%, and only in a few reports has it reached 10% (de Klein et al., 2001). Based on a N mass balance approach, Hristov et al. (2011b) estimated that more than 25% of the feed N input on a dairy farm was not accounted for in milk and manure after 24 h following excretion and this was mostly attributed to NH_3 volatilization; losses
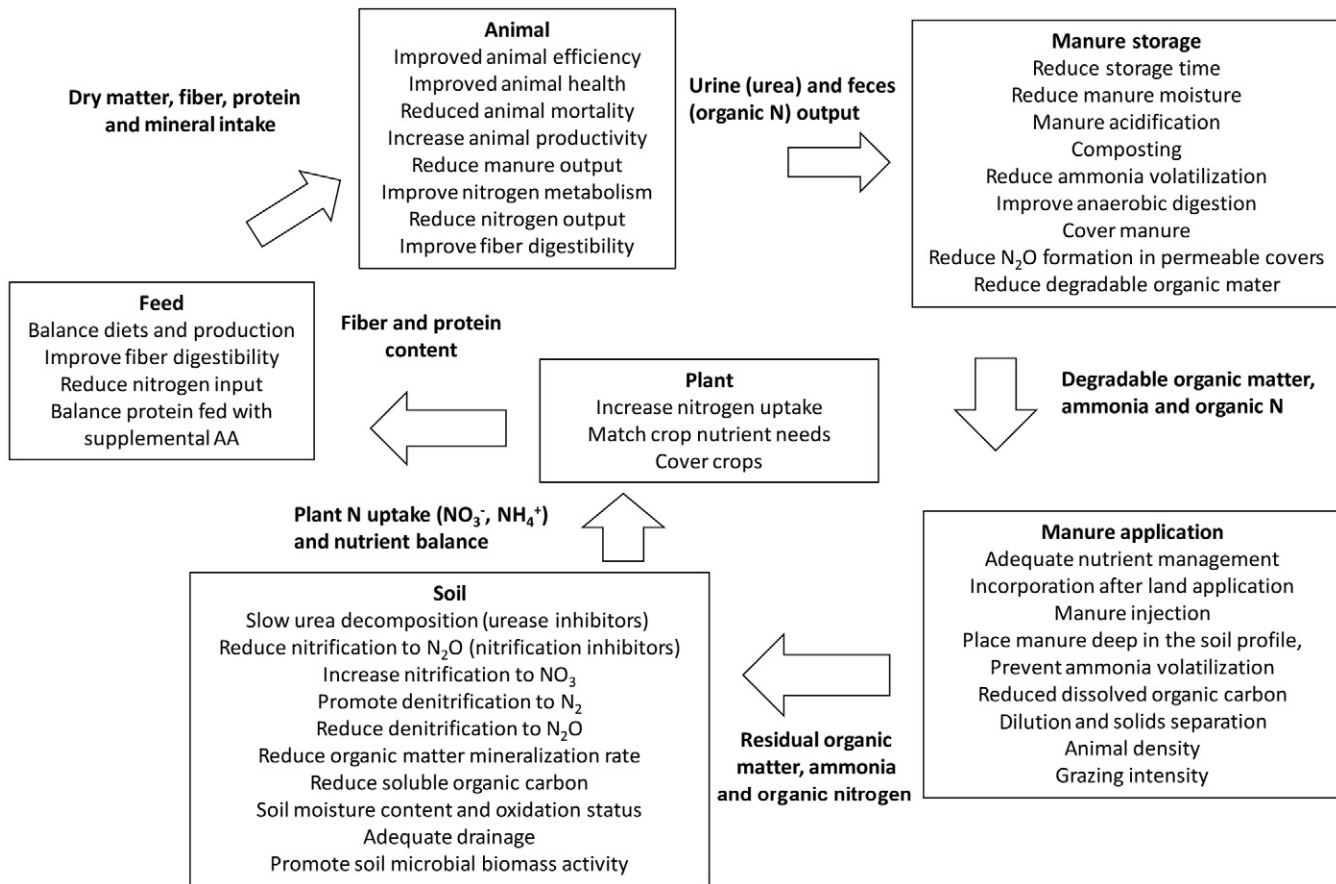
from beef cattle feedlots with long term exposure were even greater, reaching 50% (Cole and Todd, 2009).

The relationship between manure NH_3 volatilization and N_2O emission is also complex because 1) emissions of both may be reduced by diet manipulation or manure management and 2) if a mitigation technology reduces NH_3 losses, the preserved NH_4^+ may later increase soil N_2O emissions (Petersen and Sommer, 2011). On the other hand, gaseous losses of N will reduce the availability of N for nitrification and denitrification processes and, consequently, N_2O formation (USEPA, 2010). Therefore, NH_3 emission is considered an important component of the analysis of N_2O mitigation practices presented in this review.

ANIMAL MANAGEMENT AND HOUSING

Animal Housing. Structures used to house livestock do not directly affect the processes resulting in N_2O and CH_4 emissions; however, the type of structure used determines the manure management methods used to handle, store, process, and use the manure. Housing systems with solid floors that use hay or straw for bedding accumulate manure with higher DM, which is commonly

Opportunities to mitigate nitrous oxide from livestock manure

Animal
Improved animal efficiency
Improved animal health
Reduced animal mortality
Increase animal productivity
Reduce manure output
Improve nitrogen metabolism
Reduce nitrogen output
Improve fiber digestibility

Dry matter, fiber, protein and mineral intake

Urine (urea) and feces (organic N) output

Manure storage
Reduce storage time
Reduce manure moisture
Manure acidification
Composting
Reduce ammonia volatilization
Improve anaerobic digestion
Cover manure
Reduce N_2O formation in permeable covers
Reduce degradable organic mater

Feed
Balance diets and production
Improve fiber digestibility
Reduce nitrogen input
Balance protein fed with supplemental AA

Fiber and protein content

Plant
Increase nitrogen uptake
Match crop nutrient needs
Cover crops

Degradable organic matter, ammonia and organic N

Plant N uptake (NO_3^-, NH_4^+) and nutrient balance

Soil
Slow urea decomposition (urease inhibitors)
Reduce nitrification to N_2O (nitrification inhibitors)
Increase nitrification to NO_3
Promote denitrification to N_2
Reduce denitrification to N_2O
Reduce organic matter mineralization rate
Reduce soluble organic carbon
Soil moisture content and oxidation status
Adequate drainage
Promote soil microbial biomass activity

Residual organic matter, ammonia and organic nitrogen

Manure application
Adequate nutrient management
Incorporation after land application
Manure injection
Place manure deep in the soil profile,
Prevent ammonia volatilization
Reduced dissolved organic carbon
Dilution and solids separation
Animal density
Grazing intensity

Figure 2. The flow of nitrogen and organic carbon through the livestock production system including opportunities to mitigate nitrous oxide from livestock manure.

stored in piles creating conditions conducive for nitrification and denitrification and thus greater N_2O emission. Külling et al. (2001, 2003) compared liquid manure with stacked manure handling systems, and their results indicate that farm yard manure and deep litter manure handling systems tend to produce greater N_2O emissions than slurry-based systems. In these studies, quantitative differences in N_2O emissions from the manure handling systems were difficult to determine because protein content in the diet and NH_3 emission from manure also varied. Greater CH_4 emissions were reported from farm yard manure followed by liquid slurry and deep litter manure. Amon et al. (2001) compared composted, anaerobically stacked, and slurry-based manure and found higher NH_3 emissions in composted manure with most of the losses occurring after manure was turned during aeration. These authors found much greater N_2O and CH_4 emissions from anaerobically stacked manure with no significant difference between slurry-based and straw-based manure systems.

Housing systems with slatted floors accumulate manure in liquid or slurry form with that manure stored for longer periods of time increasing the production of CH_4

and reducing the production of N_2O. Hassouna et al. (2010) studied gaseous emissions from cattle housing in France and found higher N_2O emissions in buildings with straw-based bedding and solid manure handling systems when compared with liquid manure handling systems. Nitrous oxide emissions were detected in only 2 of the 14 liquid manure systems studied. The same study found smaller differences between CH_4 emissions from buildings using straw-based solid manure or liquid manure systems and attributed this result to the difficulty in discerning enteric CH_4 emissions from manure emissions because the former produced most of the CH_4 emitted from the buildings.

Hristov et al. (2012) investigated the effect of manure management on barn floor NH_3, CH_4, N_2O, and CO_2 emissions from 12 commercial dairy farms in Pennsylvania. Dairies participating in the study had flush (manure was flushed twice daily), 2 types of scrape (manure was scraped daily), and gravity-flow (manure was accumulated under the building and removed several times during the year) manure systems. Barn floor NH_3 emissions were considerably lower for the flush manure systems (average of 167 mg/m² per h) and highest for the gravity-flow system (426 mg/m² per h). Methane emissions were also

lowest for the flush (37 mg/m^2 per h) and much higher for the gravity-flow system (1,216 mg/m^2 per h). Carbon dioxide emissions were not different among manure systems (ranging from about 2,000 to 7,000 mg/m^2 per h), and N$_2$O emissions were negligible in all systems. This study showed that NH$_3$, and particularly CH$_4$, emissions from manure are much greater from dairy barns where manure is stored for prolonged periods of time compared with barns where manure is removed daily.

In contrast to ruminants, housing plays a more important role on GHG mitigation in nonruminant livestock production systems because most of the emission in these systems comes from the manure. Philippe et al. (2007) compared GHG emissions from fattening swine raised on a concrete slatted floor or straw-based deep litter. Swine fattened on deep litter released nearly 20% more GHG than those on slatted floors (6.2 and 13.1 g/swine per day for NH$_3$, 0.54 and 1.11 g/pig per day for N$_2$O, and 16.3 and 16.0 g/swine per day for CH$_4$, respectively).

The type of housing system also determines the feasibility of using anaerobic digestion or composting to treat the manure with its associated effects on GHG emissions. As described by the International Atomic Energy Agency (IAEA, 2008), the types of housing used in Asia facilitate different strategies for manure treatment depending on the capital resources of the producer. A popular alternative for smallholders is anaerobic digestion of animal and household waste, usually funded by government programs. Medium to large producers have better access to capital investment, have specialized production determined by surrounding markets, and generally use raised slatted floors that allow collection of manure for further treatment through solids separation and anaerobic digestion.

Mechanically ventilated structures provide opportunity to treat emitted GHG through filtration and scrubbing as gases are exhausted from facilities. An interesting mitigation technology for animal housing uses titanium dioxide (TiO$_2$) paint on the interior walls. Industrial uses of TiO$_2$ show that stimulation of its photocatalytic properties by UV light leads to oxidation of NH$_3$ and NO$_x$ (e.g., Lee et al., 2002; Allen et al., 2005). Studies by Guarino et al. (2008) and Costa et al. (2012) in swine houses showed that GHG mitigation with TiO$_2$ paint holds promise. Alkali and alkaline earth metal oxides, hydroxides, and carbonates or bicarbonates have been shown to have high CO$_2$ absorption capacity and are being investigated for CO$_2$ sorbent applications (Duan et al., 2012).

Dietary Effects on Manure Emissions. Diet manipulation to reduce nutrient excretion has been studied for many years, especially related to N and P reduction, but its consideration as a mitigation practice to reduce CH$_4$ and N$_2$O emissions from manure is relatively new. Diet can have a profound effect on N losses and particularly

the route of N excretion (i.e., feces vs. urine) in most farm animals (Hristov et al., 2013b). Studies with ^{15}N-labeled urine or feces have demonstrated that urinary N is the primary source of NH$_4^+$ in cattle manure, contributing from 88 to 97% of the NH$_3$ emitted within the first 10 d of manure storage (Fig. 3; Lee et al., 2011a).

With urine being the main source of volatile N emissions, manipulating the route of N excretion becomes an important N$_2$O and NH$_3$ mitigation tool. Urea is the main nitrogenous constituent of ruminant urine. In the urine of high-producing dairy cows, urea N represents 60 to 80% or more of total urinary N (Reynal and Broderick, 2005; Vander Pol et al., 2008) and proportionally decreases as dietary CP and intake decrease (Colmenero and Broderick, 2006). On low-protein diets, urinary urea N can be as low as 46 to 53% of the total urinary N (Hristov et al., 2011a; Lee et al., 2012a) with urinary N close or even below 19% of total excreta N (Lee et al., 2011b, 2012b). Therefore, reducing dietary CP concentration is perhaps the most effective method for mitigating NH$_3$ emissions from stored manure (Hristov et al., 2011b). Emissions from land-applied manure are further reduced because low-CP diets produce manure with a slower mineralization rate of N (Powell and Broderick, 2011). Sauvant et al. (2011) showed, however, that CH$_4$ production per kilogram digested OM decreased in a linear fashion with increasing dietary CP, that is, decreasing dietary protein concentration likely results in increased concentration of fermentable carbohydrates in the diet, which in turn likely increases CH$_4$ production. In support of this, Dijkstra et al. (2011) concluded, from a simulation study, that dietary-N mitigation options at the animal level aimed at reducing urinary N excretion may result in elevated enteric CH$_4$ emission (per kg of fat- and protein-corrected milk). These potential relationships must be considered when manipulating dietary N to reduce manure NH$_3$ and N$_2$O emissions.

Several studies have investigated the effect of dietary protein on N$_2$O (and CH$_4$) emissions from manure and manure-amended soil. Külling et al. (2001) reported decreased N$_2$O emissions during simulated storage of manure from dairy cows fed low-protein diets, but the total GHG emissions were not affected by the dietary protein content (due to increased CH$_4$ emissions from the low-protein manure). Velthof et al. (2005) concluded that decreasing the protein content of swine diets had the largest potential to simultaneously decrease NH$_3$ and CH$_4$ emissions during manure storage and N$_2$O emission from soil. Data on the effect of dietary protein on manure N$_2$O emissions, however, are not consistent and often no effect or even increased N$_2$O emissions (from housing) have been reported when lowering dietary protein for swine (Clark et al., 2005; Philippe et al., 2006) and dairy cattle (Arriaga et al., 2010). Manure CH$_4$ and CO$_2$ emissions per unit of

Table 1. Manure handling strategies offering non-CO_2 greenhouse gas mitigation opportunities

Category	Species[1]	Potential CH_4 mitigating effect[2]	Potential N_2O mitigating effect[2]	Potential NH_3 mitigating effect[2]	Effective[3]	Recommended[4]
Dietary strategies						
Reduced dietary protein	AS	?[5,6]	Medium	High	Yes (N_2O and NH_3)	Yes (N_2O and NH_3)
High fiber diets	SW	Low	High	?	Yes (N_2O)	Yes (N_2O)
Grazing management						
Grazing intensity[7]	AR	?	High?[7]	?[7]	Yes (N_2O)	Yes (N_2O)
Housing						
Biofiltration	AS	Low?	?	High	Yes (NH_3 and CH_4?)	Yes (NH_3 and CH_4?)
Manure system[8]	DC, BC and SW	High	?	High	Yes (CH_4 and NH_3)	Yes (CH_4 and NH_3)
Manure treatment						
Anaerobic digestion	DC, BC and SW	High	High[9]	Increase?[10]	Yes (CH_4 and N_2O)	Yes (CH_4 and N_2O)
Solids separation	DC and BC	High	Low?[9]	?[11]	Yes (CH_4)	Yes (CH_4)
Aeration	DC and BC	High	Increase?[12]	?[12]	Yes (CH_4)	Yes (CH_4)
Manure acidification	DC, BC and SW	High	?[13]	High[13]	Yes (CH_4 and NH_3)	Yes (CH_4 and NH_3)
Manure storage						
Decreased storage time	DC, BC and SW	High[14]	High[14]	High[14]	Yes (all)	Yes (all)
Storage cover with straw	DC, BC and SW	High	Increase?[15]	High	Yes (CH_4 and NH_3)	Yes (CH_4)
Natural or induced crust	DC and BC	High	Increase?[15]	High	Yes (CH_4 and NH_3)	Yes (CH_4)
Aeration during liquid manure storage	DC, BC and SW	Medium to High	Increase?[12]	?[12]	Yes (CH_4)	Yes (CH_4)
Composting	DC, BC and SW	High	?[12]	Increase[12]	Yes (CH_4)	Yes (CH_4)
Litter stacking	PO	Medium	N/A[16]	?	Yes (CH_4)	Yes (CH_4)
Storage temperature	DC and BC	High	?	High	Yes (CH_4 and NH_3)	Yes (CH_4 and NH_3)
Sealed storage with flare	DC, BC and SW	High	High	?[17]	Yes (CH_4 and N_2O)	Yes (CH_4 and N_2O)
Manure application method						
Injection vs. surface application	DC, BC and SW	No effect to increase?	No effect to increase[18]	High	Yes (NH_3)	Yes (NH_3)
Timing of application	AS	Low	High[19]	High	Yes (N_2O and NH_3)	Yes (N_2O and NH_3)
Soil cover, cover cropping	AS	?	No effect to High[20]	Increase?[21]	Yes (N_2O?)	Yes (N_2O?)
Soil nutrient balance	AS	N/A	High	High	Yes (N_2O and NH_3)	Yes (N_2O and NH_3)
Nitrification inhibitor[22]						
Applied to manure or after urine deposition in pastures	DC, BC and SH	N/A	High	N/A	Yes (N_2O)	Yes (N_2O)?
Urease inhibitor[23]						
Applied with or before urine	DC,BC and SH	N/A	Medium?	High	Yes (N_2O and NH_3)[23]	Yes (N_2O and NH_3)?
Soil management						
Soil condition, texture, water filled pore space, drainage	AR	?	Decrease or Increase[24]	?	Yes (N_2O)?	Yes (N_2O)?

[1]DC = dairy cattle; BC = beef cattle (cattle include *Bos taurus* and *Bos indicus*); SH = sheep; GO = goats; AR = all ruminants; SW = swine; PO = poultry; AS = all species.

[2]High = ≥30% mitigating effect; Medium = 10 to 30% mitigating effect; Low = ≤10% mitigating effect. Mitigating effects refer to percent change over a "standard practice," that is, study control that was used for comparison and are based on combination of study data and judgment by the authors of this document.

[3]Effectiveness is determined on the basis of greenhouse gas or NH_3 mitigation potential (in some cases, effects on feed intake and/or animal productivity were also considered).

[4]Based on available research or lack of sufficient research.

[5]? = uncertainty due to limited research or lack of data, inconsistent or variable results, or lack or insufficient data on persistency of the effect.

[6]Insufficient research. Modeling suggests that enteric CH_4 may increase. If rumen function is impaired, manure CH_4 emissions may increase.

[7]Reduced grazing intensity can reduce up to 50% N_2O emissions by lowering urinary N input to the soil but can also increase N_2O emissions by increasing residual OM during soil freeze-thaw cycles. Ammonia emissions may increase; magnitude is unclear.

[8]Generally, a manure system that reduces the time between feces and urine excretion and removing manure from the animal house will reduce NH_3 and CH_4 emissions from the building (data on N_2O are not conclusive and there is little N_2O emission from buildings and manure storage). Type of manure system also has an effect of greenhouse gas emissions; depending on type of storage, emissions can be greater from manure that is removed daily vs. manure that resides in the animal building for longer periods of time.

[9]Anaerobic digestion and solids separation reduce the source of degradable C in the manure applied to the soil and as a consequence reduce the potential for nitrification and N_2O emissions.

[10]Ammonia emissions will increase as more organic N is decomposed to NH_3; magnitude is unclear.

[11]Reducing the solids content of manure facilitates infiltration and prevents NH_3 losses from land application.

[12](See also Aeration during liquid manure storage and Composting.) In some cases, aeration of manure may stimulate the transformation of NH_3–N into NO_3–N, which increases the potential for N_2O emissions and reduces the potential for NH_3 volatilization. In most cases, composting increases NH_3 losses.

[13]Manure acidification decreases the fraction of volatile NH_3 in manure but may increase N_2O emissions after land application.

[14]Increasing the time manure is stored increases potential emissions by increasing the concentration of NH_3 and CH_4 through degradation of nutrients in manure and the time manure emissions occur, creating a compound effect varying widely in magnitude. This strategy is listed with High mitigation potential because not storing manure would be a total avoidance of storage greenhouse gases.

[15]Covering the manure with permeable porous covers promotes nitrification and N_2O emissions.

[16]N/A = not applicable.

[17]Covering the manure storage with impermeable covers prevents NH_3 volatilization during storage and increases the content of NH_3 in the manure, which generally results in greater of NH_3 emissions after land application.

[18]Injecting manure increases the soil organic C and N pools, inducing reduction conditions and promoting N_2O emissions; the magnitude of increase depends on soil moisture, texture, temperature and many other factors.

[19]Timing land application of manure to temperature and soil moisture conditions that are not conducive to nitrification and N_2O emissions can have a large but variable impact on emissions.

[20]Cover crops can increase plant N uptake and decrease accumulation of NO_3^- and thus reduce N_2O production through denitrification, but the results on overall greenhouse gas emissions have not been consistent. Interactions with other soil conservation practices are significant (tillage system, for example) and must be considered when the goal of cover cropping is reducing whole-farm greenhouse gases emissions.

[21]Soil vegetation cover intercepts land applied manure, reducing infiltration and contact with soil, which results in greater NH_3 volatilization losses whose magnitude depends on the application method and manure characteristics.

[22]Nitrification inhibitors have reduced N_2O emissions and N leaching in intensive dairy and sheep production systems of Australia and New Zealand.

[23]Urease inhibitors work well before urea N in the urine is transformed to NH_3–N in manure; therefore, the opportunity for application may be limited to intensive feedlot production or waste handling systems, in which feces and urine are separated.

[24]Soil moisture content affects the nitrification and denitrification potential, resulting in large variation in N_2O emissions.

land may increase immediately following soil application (or during storage) due to the use of a greater application rate for low- vs. high-protein manure to meet the crops N requirements (Lee et al., 2012b).

Feeding and management can significantly affect N excretions and volatilization losses from beef feedlots. Phase feeding is one example of an effective mitigation practice for these types of production systems. Reducing dietary protein concentration during the production cycle to better meet the requirements of the animal can significantly lower N excretions (Cole et al., 2005, 2006; Vasconcelos et al., 2007) and consequently losses from the pen surface. Erickson and Klopfenstein (2010) reported 12 to 21% lower N excretion and 15 to 33% lower N volatilization losses of phase-fed cattle. Manure management can also have a significant impact on N losses. Pen cleaning frequency, for example, decreased N volatilization losses by 19 to 44% and increased manure N by 26 to 41% (Erickson and Klopfenstein, 2010).

Figure 3. Proportion of ammonia N emitted from manure originating from fecal N and urinary N. In this experiment feces or urine were labeled with ^{15}N through a continuous infusion of $^{15}NH_4Cl$ into the rumen of lactating dairy cows (from Lee et al., 2011a).

Reduced protein N in the animal diet produces manure with a slower N mineralization rate that releases less plant-available N (Powell and Broderick, 2011). Therefore, changes in manure application rate recommendations are needed to reflect N cycling from modified diets. At equal N application rates, whole-crop barley yield was not different between manures from dairy cows fed high- (16.8) or low-crude protein (14.8%) diets (Lee et al., 2013). To minimize N_2O production in all cases, manure application rates should be coordinated with the amount of mineral fertilizer applied, and consideration should be given to application timing and method to prevent N application in excess of plant requirements.

Low-protein diets must be formulated to meet or exceed the animal's energy, metabolizable protein, and AA requirements if feed intake and animal performance are to be maintained (Lee et al., 2011b). Diets severely deficient in RDP will reduce total tract fiber digestibility in ruminants, which may negatively affect DMI and animal performance (Mertens, 1994; Lee et al., 2011b; Aschemann et al., 2012). A meta-analysis by Nousiainen et al. (2009) and Huhtanen et al. (2009) showed that diet CP was the only dietary factor (of the factors studied in that analysis) that was positively related to NDF digestibility in dairy cows. Decreased ruminal degradability of fiber will increase excretion of fermentable OM in manure, which might increase manure CH_4 emissions although the latter effect has not been consistently reported (Hindrichsen et al., 2005). On the other hand, these effects may be counteracted by reduced enteric CH_4 production because fiber degradability in the rumen will decrease. Diets severely deficient in RDP will have a negative impact on microbial protein synthesis and animal productivity and therefore must not be recommended as a mitigation practice.
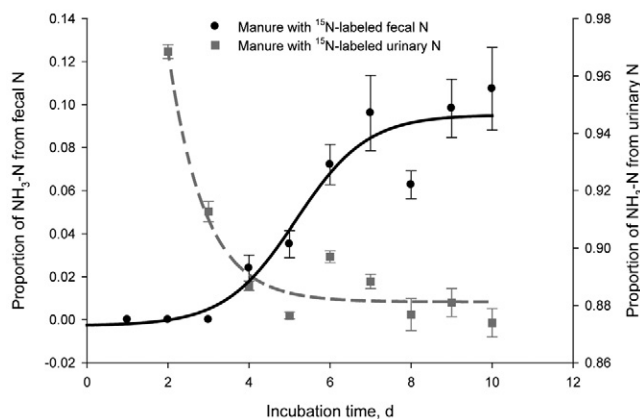
Feed intake depression with protein- and AA-deficient diets has also been demonstrated with pigs and poultry (Henry, 1985; Picard et al., 1993) and must be avoided to maintain efficient animal production. Supplementation of low-protein diets with synthetic AA may alleviate undesirable effects on feed intake. Growing swine fed a 14% CP diet containing supplemental lysine (0.73% Lys) had intake and growth performance similar to swine fed a 16% protein diet (0.77% Lys; Baker et al., 1975). Analogous results were reported by Yen and Veum (1982) who observed feed intake and ADG for growing swine fed a protein deficient (13% CP) diet supplemented with Lys and Trp to be similar to swine fed a 16% CP diet. Similar to monogastric animals, supplementation of the diet with rumen-protected AA (Lys, Met and His) increased DMI and milk production in high-producing dairy cows (Lee et al., 2012a). Overall, feeding protein close to animal requirements, including varying protein concentration with the productive stage of the animals (phase feeding), is recommended as an effective manure NH_3 and N_2O emission mitigation practice.

De Klein and Eckard (2008) concluded that abatement of N_2O should be considered as part of an integrated approach to improve the efficiency of N cycling in animal production systems. Particular attention should be given to improving animal N utilization, thus reducing urinary N output to the soil–plant system. According to these authors, current technologies could deliver up to 50% reduction in N_2O emissions from an animal housing system but only up to 15% from a grazing system. Dietary CP reduction can reduce both CH_4 and N_2O emissions from stored manure (Atakora et al., 2011a,b; Osada et al., 2011) as well as following land application (Velthof et al., 2005). Land application reductions are not supported by all data because of the large variation in soil conditions. In intensively managed pastoral systems, supplementation of the pasture with low-N feeds such as corn or small grain silage, which generally reduce dietary N concentration, can reduce urinary N losses and, consequently, manure and soil NH_3 and N_2O emissions (by 8 to 36%; de Klein and Monaghan, 2011). In some systems, however, this reduction may be of a smaller magnitude (Velthof et al., 2009), or total GHG emissions may be even increased (Beukes et al., 2010), perhaps due to increased synthetic fertilizer use to grow the cereal silage.

Shifting N losses from urine to feces is expected to reduce N_2O emissions from manure-amended soil due to the lower concentration of NH_4^+ in manure (this will depend on storage conditions, if manure is stored before application). Feed additives, such as tannins, have redirected excreted N from urine to feces. Carulla et al. (2005), for example, reported a 9.3% reduction in urinary N as proportion of total N losses, and Misselbrook et al. (2005) reported a reduction in manure NH_3 emissions

from cows fed tanniferous forage (although the effect in the latter study was confounded by the CP content of the diets). Grainger et al. (2009) observed a 45 to 59% reduction in urinary N excretion (as percent of N intake) with condensed tannins but also a 22 to 30% drop in milk N secretion. Similarly, Aguerre et al. (2010) observed a linear decrease in urinary N excretion (vs. a linear increase in fecal N excretion) in high-producing dairy cows fed diets supplemented with 0 to 1.8% (DM basis) of a quebracho tannin extract. Ammonia emission from slurry from cows receiving the tannin-supplemented diets was 8 to 49% lower than emissions from the control slurry. Tannins also reduced NH_3 emission by 20% when directly applied to the barn floor and 27% after a tannin extract was applied to soil (Powell et al., 2011a,b).

Studies directly investigating the effect of tannins on manure or soil N_2O emissions are scarce. Hao et al. (2011) supplemented cattle diets with condensed tannins from *Acacia mearnsii* at 25 g/kg DM and followed GHG emissions from composted manure for up to 217 d. Nitrous oxide emissions that occurred during the first 56 d of composting were generally low (up to 0.1 kg N/t compost DM) and not affected by tannin supplementation. Methane (and CO_2) emissions were also not affected by tannin supplementation. The authors speculated that tannin application level was too low, tannins were complexed with protein on excretion in the feces, or that microbes in compost were capable of altering the biological activity of tannins.

Decreased N release rate from manure from animals fed tanniferous forages has been reported (Powell et al., 1994; Cadisch and Giller, 2001) although other reports indicated no effect of condensed tannins on the agronomic value of cattle manure (Hao et al., 2011) and decreased manure N availability may be a concern in agricultural systems relying exclusively on manure as a source of N for crop growth. Indeed, some reports have indicated a significant drop in yield when high-tannin manure (equivalent to 2.2 t tannins/ha per yr) was added to sweet corn and radish plots (27 to 32% reduction for sweet corn and 42 to 46% for radish; Ingold et al., 2012). However, others reported no effect of condensed tannins on the N fertilizer value of feces from sheep receiving a diet supplemented with tanniferous legumes (Tiemann et al., 2009). Clearly, more studies are needed to relate tannin application, through the diet or directly to manure, to GHG emissions from manure during storage or after land application.

Grazing Practices. Improving pasture quality in terms of forage digestibility is an efficient way of decreasing GHG emissions from the animal and the amount of manure produced. However, in pasture-based production systems, improving forage quality often means increasing N fertilizer application rates,

which can have a negative impact on urinary N excretion and thus NH_3 and N_2O emissions.

Nitrous oxide emissions can be particularly high in intensive pasture systems due to high N concentration in urine as a result of the high CP content of pasture (22 to 28% CP, DM basis, in New Zealand, for example), and there are many reports on the relationship between the placement and chemical composition of urine and soil nitrification and denitrification processes. Eckard et al. (2010) pointed out that the effective N application rate within a urine patch from a dairy cow on pasture is between 800 and 1,300 kg N/ha, and N is deposited at concentrations that are orders of magnitude greater than the utilization capacity of the soil–plant system. These authors suggested that a more uniform distribution of urine throughout the paddock would reduce the effective N application rate, which should translate into a reduction in N_2O emissions.

These effects are often compounded by high fertilizer N application rates to stimulate grass growth, which further increases urinary N concentration. De Klein et al. (2001) showed a 40 to 57% reduction in N_2O emissions when grazing was restricted to 3 h/d in the humid late autumn in New Zealand. This reduction was attributed to diminished N input during conditions most conducive to N_2O emissions in New Zealand. When de Klein et al. (2001) included N_2O emissions resulting from application of the effluent collected during restricted grazing periods, N_2O emissions were reduced by only 7 to 11%.

Nevertheless, keeping the animals off the paddocks, in "stand-off" or "feed pads" for most of the day during the wet months of the year (autumn–winter), has been shown to be an effective N_2O mitigation practice in intensive grazing systems (de Klein, 2001; de Klein et al., 2002; Luo et al., 2008a). Not allowing grazing during wet weather also increases pasture productivity due to reduced sward damage and soil compaction (de Klein, 2001; de Klein et al., 2006). One must keep in mind, however, that this practice results in much greater NH_3 emissions (Luo et al., 2010) due to urine and feces being excreted and allowed to mix in the stand-off or feed pad area. According to Luo et al. (2010), reduction of N_2O emissions from intensive grazing systems can be achieved by several strategies: 1) improving N use efficiency through reducing the amount of N excreted by grazing animals, 2) optimizing soil management and N inputs, 3) optimizing pasture renovation, 4) manipulating soil N cycling processes through soil additives, 5) selecting for plants and animals that maximize N utilization, and 6) altering grazing and feeding management.

Biofiltration. Biofiltration can treat ventilated air from animal buildings using biological scrubbers to control odor, to absorb NH_3, and to convert NH_3 into NO_3. Preventing NH_3 losses may indirectly reduce N_2O emissions by reducing NH_4^+ deposition and consequent conversion to N_2O (see earlier discussion).

Ammonia removal efficiency in swine and poultry houses from acid scrubbers and biotrickling filters (based on biofilms that degrade the odorous compounds) averaged 96 and 70%, respectively (Melse and Ogink, 2005). Shah et al. (2011) investigated the effectiveness of a coupled biofilter–heat exchanger in reducing NH_3 emissions (and recover heat) in a broiler house. The biofilter was effective in treating very high inlet NH_3 concentrations (>96 mg/kg) with removal efficiencies greater than 79% for empty bed residence times ranging from 4.3 to 29.1 s. The biofilter was apparently also able to trap some sulfurous gases emitted from the broiler house.

Recent reports (Maia et al., 2012a,b) have shown that biofilters used to scrub NH_3 from exhaust streams in animal houses generate N_2O as a result of nitrification and denitrification processes in the biofiltration media. In their first study, Maia et al. (2012a) showed a high correlation between biofilter NH_3 removal and N_2O generation associated with conversion of NH_3 to NO_2^- and NO_3^- in the biofilter. In their second study, Maia et al. (2012b) reported that moisture content between 48 and 52% in the biofilter media was an important factor in obtaining significant NH_3 reduction with reduced N_2O production. Therefore, N_2O production in biofilter scrubbers should be taken into account when implementing biofiltration systems for GHG and NH_3 mitigation.

A few studies have investigated CH_4 mitigation by passing contaminated air from a swine manure storage or from swine housing through a biofiltration system. A Canadian Pork Council (2006) study reported reductions of 50 to 60%, and Girard et al. (2011) reported a maximum reduction of up to 40%. They described their biofilter as "packed with inorganic material," but the packing material was not disclosed.

Melse and van der Werf (2005) reported up to 85% CH_4 removal from the exhaust stream of a covered swine liquid manure storage using a biofiltration system composed of a mixture of compost and perlite inoculated with CH_4 oxidizing bacteria collected from activated sludge. The CH_4 removal capacity of the biofilter system depended on the concentration of CH_4 in the filtered stream. Therefore, the authors extrapolated that an equivalent system for animal housing with low CH_4 concentration in the filtered stream would require very large biofilter systems to achieve 50% reduction or more, pointing to this as a limitation in the applicability of this technology. In addition, they reported N_2O production in the biofiltration system contributing 4 to 64% of the outlet stream GHG CO_2e, which, as mentioned above, needs to be included in the design and promotion of biofiltration strategies for GHG mitigation. The high residence time necessary in these systems due the low

solubility and biodegradability of CH_4 hinders effectiveness (Melse and Verdoes, 2005). Melse and Timmerman (2009) reported on the potential use of multipollutant scrubbers, combining acid scrubbers, biofilters, and water curtains to reduce not only NH_3, odors, and GHG but also particulate matter from animal housing exhaust.

MANURE STORAGE AND TREATMENT

Greenhouse gas emissions from stored manure are primarily in the form of CH_4 (due to anaerobic conditions) although N_2O emissions can occur and NH_3 volatilization losses are often large. A direct way to avoid cumulative GHG emissions is to reduce the time manure is stored (Philippe et al., 2007; Costa et al., 2012). Increasing the time of manure storage increases the period during which CH_4 (and potentially N_2O) is emitted as well as the emission rate, creating a compound increase (Philippe et al., 2007).

Temperature is a critical factor regulating processes leading to NH_3 (Sommer et al., 2006) and CH_4 (Steed and Hashimoto, 1994) emissions from stored manure. Decreasing manure temperature to <10°C by removing the manure from the building and storing it outside in cold climates can reduce CH_4 emissions (Monteny et al., 2006).

Storage treatments with proper aeration and moisture management have reduced CH_4 generation from poultry manure (Li and Xin, 2010). Ventilated belt removal of laying hen manure can reduce CH_4 emissions compared to deep-pit storage (Fabbri et al., 2007). Separation of swine slurry into solid and liquid portions and then treating the solids through aerated composting reduced CH_4 emissions by 99% and N_2O emissions by 75% compared with untreated manure (Vanotti et al., 2008). However, due to the often negative relationship between NH_3 and N_2O emissions (Petersen and Sommer, 2011), this process is likely to increase NH_3 emissions substantially and perhaps total N losses from manure. Amon et al. (2001) reported greater NH_3 losses from an actively turned composting pile of solid cattle manure than from an undisturbed anaerobically stored pile, with the opposite effect observed for N_2O emissions.

Prapaspongsa et al. (2010a,b) compared 14 swine manure management practices based on combinations of thermal pretreatment, anaerobic digestion, anaerobic co-digestion, liquid and solid separation, drying, incineration, and thermal gasification with respect to their energy, nutrient, and GHG balances. The anaerobic digestion-based scenario with a natural crust during storage had the highest GWP reduction through high efficiencies in energy and nutrient recovery with restricted emissions of GHG and NO_3^-. The incineration and thermal gasification-based scenarios and a scenario using only deep injection yielded the greatest reduction in respiratory inorganics and terrestrial eutrophication categories because they had the lowest NH_3 emissions. Manure incineration combined with liquid and solid separation and drying of the solids was a promising management option yielding a high potential energy utilization rate and GHG reduction.

Storage Covers. Several types of manure storage covers have been reported in the literature including natural crusts on slurry manure stored with a high solids content, straw, wood chips, oil layers, expanded clay pellets, wood, and semipermeable and sealed plastic covers. The effectiveness of the cover depends on many factors, including permeability, cover thickness, degradability, porosity, and management. Semipermeable covers tend to increase N_2O emission because they provide optimal aerobic conditions for nitrification at the cover surface and at the same time create a low oxygen environment just below the cover favorable for denitrification and the production of N_2O (Hansen et al., 2009; Nielsen et al., 2010). Semipermeable covers are valuable for reducing NH_3, CH_4, and odor emissions, but they often increase N_2O emissions (Sommer et al., 2000; Guarino et al., 2006; VanderZaag et al., 2008).

Capturing the gases produced using impermeable membranes, such as oil layers and sealed plastic covers, can reduce NH_3, N_2O, and CH_4 emissions. The results from Guarino et al. (2006) and VanderZaag et al. (2008) suggest that using a vegetable oil layer as a manure storage cover, although very effective, is not practical because of degradability, generation of foul odors, and difficulty in preventing the oil film from becoming mixed or "broken" over the manure surface.

Covering manure storages with impermeable covers is an effective mitigation practice if the CH_4 captured under the cover is burned using a flare or an engine generator set to produce electricity; otherwise, the captured CH_4 builds pressure inside the storage creating an explosion hazard and/or rupture of the cover. Increased air pressure inside the storage structure reduces the fraction of compounds in the gas phase and increases that trapped in liquid manure. The increased gas trapped in the manure liquid is then released when the pressure in the manure storage is reduced to remove the manure. Retaining the CH_4 produced is not beneficial if it escapes at a later stage; therefore, burning or combusting the collected CH_4 to produce electricity or heat is the most desirable option. The effectiveness of impermeable covers depends on transforming the collected gases to less potent GHG such as NO_x and CO_2 (Nicolai and Pohl, 2004; Rotz and Hafner, 2011).

Anaerobic Digestion. Anaerobic digestion is the process of degradation of organic materials by archaea in the absence of oxygen, producing CH_4, CO_2, and other gases as by-products. This provides a promising practice for mitigating GHG emissions from collected

manure. In addition, when correctly operated, anaerobic digesters are a source of renewable energy in the form of biogas, which is 60 to 80% CH_4, depending on the substrate and operation conditions (Roos et al., 2004). Anaerobic digesters also provide opportunities to reduce pathogens and manure odor.

Three practical temperature ranges are generally considered for anaerobic biogas systems: psychrophilic (15 to 25°C), mesophilic (30 to 38°C), and thermophilic (50 to 60°C). These temperature ranges facilitate the growth of specific microbes. Thermophilic systems are more sensitive to environmental changes, such as temperature fluctuations and chemical concentrations produced during the digestion process (Kim et al., 2002; Ahn and Forster, 2002; El-Mashad et al., 2003), because the number of functional microorganism species that thrive at this temperature is considerably less than those that survive at lower temperatures (Ziekus, 1977; Wolfe, 1979; Smith, 1980). Below 15°C, the production of biogas is greatly reduced and CO_2 becomes the dominant product of anaerobic digestion; therefore, anaerobic digestion systems are not recommended for geographic locations with average temperatures below this threshold without supplemental heat and temperature control (Sommer et al., 2007).

The effluent of the digester, commonly called digestate, contains most of the soluble plant nutrients found in the feedstock and the more resilient, difficult to degrade organic material. Digestate is commonly applied directly to crops whereas the sludge, formed by precipitated minerals and undigested OM, may be composted before field application.

Digester designs vary widely in size, function, and operational parameters. Smaller digesters (6 to 10 m^3), designed to improve sanitary conditions in developing countries and to provide energy for single family dwellings, were promoted in the 1970s and 1980s throughout Asia and Latin America (Bond and Templeton, 2011; Jiang et al., 2011b). These were designed to function with the waste originating from a few animals (2 to 5 swine, 5 to 10 cows, 100 chickens, or a combination of these) together with the family dwelling waste. According to Dhingra et al. (2011), these types of digesters reduced GHG emissions from 23 to 53% when compared with households without biogas, depending on the condition of the digester, technical assistance, and operator ability. The effectiveness of these types of digesters for mitigating GHG depends mostly on the amount of CH_4 leakage that occurs through digester walls and piping delivering the biogas to the family dwelling (Dhingra et al., 2011). These small digesters have been used by farmers in developing countries only when government subsidies and economic incentives have been available (Bond and Templeton, 2011).

Commercial farm digesters are typically designed to treat liquid manures. There are 4 basic commercial farm-level anaerobic digestion vessel designs (Roos et al., 2004). The most common and simple covered lagoon digestion systems are ambient (psychrophilic) temperature systems that require manure with a solid content of 3% or less and a storage cover to maintain anaerobic conditions. These systems typically create the largest type of digester with the longest hydraulic retention time. Plug-flow digesters and fixed domes use a vessel that receives manure at one end and discharges from the opposite end with no mixing or agitation. These systems may be heated to a mesophilic temperature and require slurry with a solids content of 11 to 13%. Small-scale digesters are often of this design. The more sophisticated complete mix digesters consist of an engineered digestion vessel designed to handle manure slurries with a solid content from 3 to 10%. A mixing system enhances bacterial contact with OM. Supplemental heat is often added to these systems to operate at mesophilic temperatures, which promotes bacterial growth and a shorter hydraulic retention time. Fixed-film digesters use a medium, such as rope, plastic mesh, or beads, placed in the vessel on which bacteria can grow. Dilute manures with a solids content of 3% or less are passed across (or through) the medium in these systems. Whereas other systems rely solely on suspended microbial growth, these also feature attached microbial growth. Widespread commercial farm digester adoption has not occurred because of variable economic return (Hill et al., 1985; Safley and Westerman, 1994; Braber, 1995) and the limited competitiveness of biogas with other fuels used for heat and power (Lantz et al., 2007).

Industrial biogas digesters are used to produce renewable energy for towns and municipalities. These digesters, prevalent mostly in Europe, use biomass collected from several farms to feed the anaerobic digesters. Co-digestion of agricultural biomass, industrial organic waste, and animal manures is common in industrial biogas plants because it allows better optimization of C to N ratio and CH_4 production while reducing the impact of NH_4^+ on gas production (Ward et al., 2008).

When CH_4 is collected and used as an energy source, it can substitute for combusted fossil fuels reducing the emissions of GHG, NO_x, hydrocarbons, and particulate matter (Börjesson and Berglund, 2006). These authors compared the emissions from the life cycle of raw materials used for anaerobic digestion (6 different feedstocks, including swine manure) and the emission from systems that the anaerobic digestion process replaced. One of the serious concerns identified was uncontrolled losses of CH_4 from biogas plants, including losses from stored digestate. Typical losses from systems storing digested manure were reported to range from 5 to 20% of the total biogas produced (Bjurling and Svärd, 1998; Sommer et al., 2001).

In a follow-up report, Börjesson and Berglund (2007) further explored overall environmental impact when biogas systems replaced various energy producing reference systems. The investigation was based on Swedish conditions using an LCA approach that considered both direct and indirect emissions. Greenhouse gas emissions per unit of heat were reduced 10 to 25% when biogas-based heat replaced fossil fuel-based heat. Emissions from biogas systems contributed 60 to 75% and 25 to 40% of the life cycle emissions of CO_2 and CH_4 in the reference and fuel-based systems, respectively.

During the anaerobic digestion process, N-containing compounds found in substrates, such as proteins, AA, and urea, are reduced to NH_3 (Bernet et al., 2000). Ammonia remaining in the aqueous solution is then transferred to the soil when the digestate is land applied (Bernet et al., 2000; Hafner et al., 2006). Anaerobic digestion stabilizes the organic C in the feedstock (reducing the fraction of easily degradable C in manures) increases plant availability of N and provides less energy to support the growth of N_2O-forming microorganisms, reducing the potential of N_2O emissions when applied to soil. Mineralization of organic N and VFA during anaerobic digestion increases manure pH and available N, which may result in increased NH_3 volatilization (Petersen and Sommer, 2011).

In general, reduction of manure OM content is expected to reduce N_2O emissions from manure-amended soils (Petersen, 1999; Bertora et al., 2008) although Thomsen et al. (2010) reported higher N_2O emissions when treated manure was applied in a wet spring season. These contradictory results led Petersen and Sommer (2011) to conclude that there is not a simple relationship between removal of manure OM and the risk of N_2O emission. To address this controversy, Thomsen et al. (2010) proposed linking the balance between N_2O and N_2 to soil water-filled pore space and oxygen supply. This relationship has been discussed in detail by Petersen and Sommer (2011); the authors concluded that prediction of N_2O emissions from manure-amended soil depends on manure composition and soil conditions. Masse et al. (2011) noted high variability between N_2O emissions, referring to 6 studies that found similar differences in emissions of the gas when comparing digested and nondigested manures.

Data on the anaerobic digestion of poultry waste as a GHG mitigation practice are limited. Several studies show successful biogas production using poultry waste as a component of co-digestion (digesting poultry waste with other manures—beneficial due to the complementary composition of the different manures); however, the impact on GHG mitigation was not reported. The ability to use anaerobic digestion to create, capture, and destroy CH_4 derived from swine manure is well documented (Safley and Westerman, 1994; Masse et al., 2003a,b).

Although it is possible to reduce CH_4 emissions by over 60% from swine manure using anaerobic digestion, the amount of CH_4 produced and collected does not directly translate into an equal amount of reduced CH_4 emissions because the untreated manure would not yield the same amount of CH_4 gas. Most literature reviewed focused on research that compared digested manure with manures that received no treatment or a different treatment. In this manner, the biogas removed was not considered in the emission comparisons of nondigested versus digested manure by many authors who used the assumption that biogas produced during digestion is destroyed through controlled combustion. A number of the studies referenced here considered emissions from digested manure after it was land applied. Reductions of N_2O emissions reported in these papers were as high as 70% compared with untreated manure applications. A commonly stated reason for this decrease was that digested manure contains less OM (degradable C) providing less energy for nitrite-forming microorganisms, which subsequently limits N_2O production.

Although most anaerobic digestion systems significantly reduce GHG emissions when compared with traditional manure handling systems, incorrect operation, lack of maintenance, and CH_4 leaks can make them a net contributor to GHG. For this reason, it is imperative that system designs and components ensure containment of nearly all biogas. The potential for anaerobic digestion to mitigate N_2O emissions after the digested manure is land applied is promising, but many parameters involved with field application contribute to conflicting reports.

Anaerobic digestion systems require large initial capital investments during construction along with ongoing maintenance and supervision costs. Historically, the adoption of this technology occurs only when economic incentives are offered as price advantages for biogas (biofuels and renewable policy incentives), when the costs of construction and maintenance are subsidized, or when no competitive alternative energy source is available. Furthermore, instruction and technical assistance to users are necessary in implementing successful anaerobic digestion systems because the correct operation of anaerobic digesters is not trivial and 50% failure rates are common (Bond and Templeton, 2011; Jiang et al., 2011a).

Acidification. An important factor affecting GHG emissions, in particular NH_3, from stored manure is pH. According to Petersen and Sommer (2011), manure acidification is an effective mitigation option for NH_3 emissions, but the effect on N_2O is not well studied. The relationships between NH_3 volatilization and factors such as air velocity and turbulence, manure temperature, and manure pH have been well documented (Ndegwa et al., 2011).

Ndegwa et al. (2011) listed 15 studies in which cattle, swine, or poultry manure NH_3 emissions were success-

fully mitigated (from 14 to 100% reduction in emissions) by lowering manure pH with sulfuric, hydrochloric, or phosphoric acids, calcium chloride, alum, or monocalcium phosphate monohydrate. These authors concluded that strong acids are more cost effective at reducing manure pH than weaker acids or acidifying salts. However, strong acids are more hazardous and, therefore, acidifying salts and weaker acids may be more suitable for on-farm use. Acidification of urine and, consequently, manure from cattle or monogastric farm animals has also been attempted using anionic salts, high dietary levels of fermentable carbohydrates, organic (benzoic) acids, or Ca and P salts (Ndegwa et al., 2008).

A commercial system used on several farms in Denmark acidifies a portion of the manure with concentrated sulfuric acid to a pH of 5.5, removes a portion of the acidified manure equivalent to daily manure production, and returns the remaining manure to the storage facility (Sørensen and Eriksen, 2009). These authors concluded that NH_3 volatilization from acidified cattle and swine manure was low after both soil incorporation and surface application. Petersen et al. (2012) studied the effect of acidification on CH_4 (and NH_3) emission from fresh and aged cattle manure during 3 mo of storage using the equipment described above. Manure pH was adjusted to 5.5 with sulfuric acid, samples of manure were stored for 95 d, and NH_3 and CH_4 emissions were monitored. Manure pH increased gradually to 6.5 to 7 during storage. Acidification had a dramatic effect on emissions, reducing CH_4 by 67 to 87% (more pronounced with aged manure) and almost completely eliminating NH_3 emissions. The authors concluded that manure acidification may be a cost-effective GHG mitigation practice.

Application of acidified manure is not expected to greatly impact crop production; the pH range of acidified manure is within the optimal range for corn and many cereal crops (5.5 to 6.5; Tisdale et al., 1993). Approximately 30% of soils worldwide and about 60% in Asia are acidic (<pH 5.5) and already require periodic lime applications to maintain optimal pH (von Uexküll and Mutert, 1995). Smaller quantities of acidified manure would be needed to provide crop N requirements because the reduction in NH_3 emissions provides manure with a greater plant-available N content. However, long-term impacts of land application of acidified manures on soil pH have not been reported, and more frequent application of lime to maintain optimal pH in some soils may be required.

Composting. Composting is an exothermic, aerobic process of microbial decomposition of OM that has several benefits related to manure handling, odor control, pathogen control, OM stabilization, additional farm income, etc. Composted manure solids (following manure separation into solids and liquid) are also used as bed-

ding in some dairy production systems to reduce the cost of production and provide cow comfort, assuming udder health is not compromised (Husfeldt et al., 2012).

Due to the nature of the composting process, N losses can be high and are influenced by a number of factors including temperature, C to N ratio, pH, moisture, and material consistency (Zeman et al., 2002). Compost can be a source of N_2O emissions with both nitrification and denitrification processes occurring during composting. *Bacillus* species are the main players in the degradation of OM and betaproteobacterial NH_3–oxidizing bacteria involved in the nitrification process (Maeda et al., 2011). Depending on composting intensity, NH_3 losses can be particularly high, reaching 50% of the total manure N (Peigné and Girardin, 2004). Aeration of the composting heap reduces CH_4 emissions (Thompson et al., 2004; Jiang et al., 2011a; Park et al., 2011) but can increase NH_3 and N_2O losses (Jiang et al., 2011a). Hao et al. (2004) reported up to 30% DM, 53% C, and 42% of initial N being lost during composting of straw-bedded manure. Methane losses accounted for 6% of the C losses. Nitrous oxide losses represented 1 to 6% of the total N losses.

Addition of mature compost with nitrite-oxidizing bacteria to actively composting swine manure was shown to reduce N_2O emission by 70% (Fukumoto and Inubushi, 2009). These authors reported that up to 19% of the total manure N was lost as NH_3 and N_2O. Brown et al. (2008) reviewed the impact of composting of a range of feedstocks (including animal manure) on GHG emissions and pointed out that the primary benefit of composting is the reduction of CH_4 emissions compared with manure stored under anaerobic conditions. These authors estimated, for example, that a facility that composts an equal mixture of manure, newsprint, and food waste could conserve the equivalent of 3.1 Mg CO_2 per Mg of dry feedstock composted if feedstocks were diverted from anaerobic storage lagoons and landfills without gas collection mechanisms.

According to Clemens et al. (2006), raw cattle manure can release from 160 (winter) to 3,600 (summer) g/m^3 of CH_4 and 38 to 57 g/m^3 of N_2O. For digested manure, the release rates are from 80 (winter) to 1,200 g/m^3 (summer) CH_4 and 40 to 76 g/m^3 N_2O, respectively. A recent study by Kariyapperuma et al. (2012) reported a 57% decrease in soil N_2O emissions with composted vs. liquid swine manure. Remarkably, emissions during the same period of the following year were not different between composted and noncomposted manure; the authors attributed the lack of difference to a significant reduction in emissions in the second year due to frozen soil. In spite of significant GHG emissions from composting, the review by Brown et al. (2008) concluded that even in a worst-case scenario, these emissions are minimal in comparison to the benefits associated with

the CH_4 reduction credits from composting. The authors also stated that it is possible to significantly reduce emissions from compost piles by increasing the solids content of the feedstocks and the C to N ratio. Overall, Brown et al. (2008) concluded that composting can be an effective method for reducing GHG emissions from a range of waste materials, including animal manure. It must be noted, however, that NH_3 losses during manure composting are significant.

LAND APPLICATION

Manure is a valuable resource that is best used as fertilizer. However, increased animal density accompanied by continuous inflow of nutrients from imported feeds can lead to nutrient imbalance at the farm and watershed scales, which can lead to greater water and air pollution. This nutrient imbalance is more likely to occur in intensive animal production systems. When input of recoverable manure nutrients (the quantity of manure nutrients available for land application) grossly exceeds the assimilative capacity of soil and crop utilization, nutrient buildups occur (Saam et al., 2005). Lander et al. (1998) categorized the ratio of recoverable manure nutrients to the assimilative capacity of crop and pasture land at a county level from 1.0 (indicating that the county had county-level excess nutrients) to <0.25 (less than 25% of the nutrients taken up and removed by crops or applied to pasture can be supplied by manure generated within the county). According to the USDA (Kellogg et al., 2000), the number of counties in the United States in 1997 with ratios of 0.5 or greater (i.e., surplus of nutrients) totaled 165 for N and 374 for P. Maguire et al. (2007) estimated that in 89% of counties in the United States, there was a deficit of manure P relative to crop P removal and there was a manure P surplus in the remaining 11%. Therefore, in some regions, manure application can be limited by soil accumulation of nutrients.

Surplus nutrients can be a significant environmental problem for large livestock operations. For example, Hristov et al. (2006) reported average efficiency of use of imported N and P (total exports/total imports × 100) on commercial dairy farms in Idaho (average size of 2,100 cows and 186 ha arable land) of 41 and 66%, respectively. Accumulation of nutrients was occurring on these dairies and as a result, soil P levels in the top 30-cm layer were well above state threshold standards. Soil NO_3^- N was >40 mg/kg for 5 of the 8 dairies, and 2 were over 80 mg/kg. Such high N concentrations exceed the crop needs for optimal growth and thus represent a high potential for N loss to the environment.

When nutrient surplus is not an issue, manure is a valuable source of available nutrients for crops, successfully replacing inorganic fertilizer. In an LCA analysis,

Adom et al. (2012) found that N fertilizer input was the largest contributor to GHG emissions for feeds used by the dairy industry in the United States: about 65% due to N_2O release on application and 35% from fertilizer manufacture. These authors recommended farmer education in fertilizer best management practices to effectively reduce GHG emissions on farms.

Application Method and Emissions. Components of the microbial biomass in the soil use CH_4 as a C source and, with the exception of rice paddies, the soil is often a CH_4 sink. Only when CH_4 concentrations exceed the metabolic capacity of the soil or when the aerobic metabolism of the soil biota is inhibited throughout the soil column are CH_4 emissions significant after land application of manure. Therefore, promoting the aerobic metabolic path and reducing CH_4 load are other approaches used to reduce CH_4 emissions after manure injection (Rodhe et al., 2006). Agricultural soils absorb on average of 1.5 kg CH_4/ha per yr (Chianese et al., 2009), but CH_4 from land-applied manures can be a source of CH_4 emission, diminishing within a few days following application. Sherlock et al. (2002) measured CH_4, NH_3, and N_2O immediately after land application of swine slurry to pasture and up to 90 d afterward. They reported high NH_3 emission rates immediately after application, decaying quickly and totaling 57 kg N/ha or 22.5% of the applied N. Methane emissions were highest immediately after manure application, coming from CH_4 dissolved in the manure. Very low emissions continued for the following week, attributed to anaerobic degradation of fatty acids in the manure. Total CH_4 emissions were slightly higher than 1 kg C/ha, which accounted for 0.08% of the C applied. In contrast, N_2O emission was initially low and dropped to background concentration levels after 90 d, but high emission peaks were observed following rainfall events. Although total N_2O emission accounted for only 2.1% of the N applied, or 7.6 kg N/ha, the authors considered N_2O to be the most important pollutant due to its greater GWP (Sherlock et al., 2002).

An important difference between mineral fertilizer and manure is that manure contains organic C, which, depending on soil conditions, may affect N_2O emissions. Manure C may increase microbial respiration rates in soil, thus depleting oxygen and providing the anaerobic conditions required for denitrification (Pelster et al., 2012). As a result, organic amendments containing large amounts of labile C and available N (cattle, swine, or poultry manure) have lead to increased soil N_2O emission compared with mineral fertilizers. An Intergovernmental Panel on Climate Change (IPCC, 2006b) report assumed N_2O emission factors (Tier 1) for mineral fertilizers and cattle, poultry, and swine manure at 1 and 2% of N input, respectively. Pelster et al. (2012) reported an N_2O emission factor for plots amended with poultry manure as 1.8% of applied N, more than double that of

the other treatments, including mineral fertilizer (0.3 to 0.9%), a result attributed to the high C content of poultry manure. These authors concluded that, compared with mineral N sources, manure application increases soil N_2O flux in soils with low C content. Soil N_2O emissions can vary greatly, and emission factors of up to 12% of N input (for nitrate-based fertilizer) and 5% for manure have been reported (de Klein et al., 2001). Nyakatawa et al. (2011) investigated CH_4 and N_2O emissions from soil receiving poultry litter or ammonium nitrate using surface soil incorporation and subsurface band application methods in conventional and no-tillage systems on a Decatur silt loam soil in north Alabama. Plots receiving ammonium nitrate were net emitters of CH_4 and N_2O whereas plots receiving poultry manure were net sinks of CH_4. Nitrous oxide emissions from manure-amended soil depended on application method; surface or soil incorporation resulted in net emission of N_2O whereas manure-amended plots under subsurface band application were net sinks of N_2O.

Incorporating manures can greatly reduce NH_3 emissions, leaving more N susceptible to emission as N_2O through nitrification and denitrification. However, reduction in NH_3 losses with incorporation means that a smaller quantity of manure is required to provide the crop N requirements, and therefore the potential for N_2O production is reduced. Subsurface injection of manure slurries into soil can result in localized anaerobic conditions surrounding the buried liquid manure, which, together with an increased degradable C pool, may result in higher CH_4 emissions than with surface applied manure. Diluting the manure or reducing the degradable C flux through solid separation or anaerobic degradation pretreatments are options to reduce CH_4 emissions from injected manure (Amon et al., 2006; Clemens et al., 2006).

A note of caution is necessary because CH_4 emissions from manure injected into soil are relatively low when compared with the reduction in NH_3 volatilization obtained through subsurface injection. Powell et al. (2011c) investigated the NH_3 volatilization mitigating potential of 3 methods of stored dairy slurry application: surface broadcast, surface broadcast followed by partial incorporation using an aerator implement, and injection. Slurry total N loss was 27.1 (20.5% as NH_3 and 6.6% as NO_3^-), 23.3 (12.0% as NH_3 and 11.3% as NO_3^-), and 9.1% (4.4% as NH_3 and 4.7% as NO_3^-), respectively. The authors reported that although slurry incorporation decreased total N loss, the conserved N did not significantly impact crop yield, crop N uptake, or soil properties at the end of the trial. They explained the lack of response to conserved N by the relatively small differences in slurry N remaining after N loss and the relatively large amount of soil N mineralization rate in the high fertility soil at the study site.

Controlling the amount of N available for nitrification and denitrification in soil as well as the availability of degradable C and soil oxidation–reduction potential are options to reduce N_2O emissions that can be achieved through the manure application method. In the first few weeks after application, manure injection often increases N_2O emission compared with surface applied manure (Dell et al., 2011). Dilution, solid separation, and anaerobic digestion pretreatments of manure before injection reduce the availability of degradable C and as a result tend to decrease N_2O emission. A number of authors have noted that wet soils tend to promote N_2O emissions and that application timing can be important. On many soils, simply avoiding application before a rain event can avoid spikes in emission rates. Maintaining soil pH above 6.5 was shown by Mkhabela et al. (2006) to help reduce N_2O emissions. Nitrous oxide emissions resulting from manure injection into soil are generally low and therefore should be weighed against the benefits of reducing NH_3 volatilization when manure is surface applied. More work is needed to further investigate overall benefits of manure application mitigation strategies.

Practices that result in increased NH_3 emissions in general will reduce the overall efficiency of the production system, reduce the amount of N being recycled on the farm, and increase the demand for N fertilizer, which could increase GHG emissions. For example, lower N_2O emissions are expected when manure is left on the soil surface compared to that incorporated into the soil, largely because a significant portion of the manure N is lost as NH_3 before undergoing nitrification and denitrification. The trade-off between reduced NH_3 volatilization and higher N_2O production may be even greater for incorporation by injection because the concentration of manure in belowground bands leads to conditions that can be more conducive to denitrification than with mixing by tillage (Dell et al., 2011). Manure incorporated in pockets in the soil through injection or shallow ditches greatly reduced NH_3 emission and resulted in reduced N losses and no CH_4 emission. Nonetheless, the increased OM in manure accelerated soil metabolism, depleting oxygen in the soil porous space and thus triggering denitrification and N_2O emissions. By using anaerobic digestion or separating manure solids, the organic content of manure is reduced, which generally results in lower emissions of N_2O after manure injection (Clemens et al., 2006; Velthof and Mosquera, 2011).

Urease and Nitrification Inhibitors. Microbial processes that result in N_2O production can be manipulated through the use of chemical additives. Urease inhibitors are effective when applied to urine before it is mixed with soil or feces. In open lot feedlots, urease inhibitors have been reported to effectively decrease NH_3 losses. For example, Varel et al. (1999) treated feedlot pens with urease

inhibitors, cyclohexylphosphoric triamide, and N-(n-butyl) thiophosphoric triamide (**nBTPT**). Whereas no urea was found in the control pens, the treated pens retained significant amounts of urea for up to 14 d following treatment. Treating the pens weekly for 6 wk further increased urea conservation, reducing NH_3 volatilization losses.

Nitrification inhibitors [the most widely used are dicyandiamide (**DCD**) and nitrapyrine] were found to reduce the amount of N_2O produced under controlled experimental or field conditions. Applied over urine and feces deposited under intensive pasture-based systems in New Zealand, nitrification inhibitors were effective in reducing N_2O emissions (de Klein et al., 1996, 2001, 2011; Di and Cameron, 2002, 2003, 2012). Luo et al. (2008b) reported up to 45% reduction in N_2O emissions from dairy cow urine applied to various soils in New Zealand by the nitrification inhibitor DCD and pointed out that the effectiveness of these compounds may be reduced under heavy rainfall. Recent national trials in New Zealand reported an average N_2O reduction by DCD of 50% (Gillingham et al., 2012).

Application of DCD has also resulted in a dramatic 68% reduction in NO_3^- leaching losses from a deep sandy soil pasture of perennial ryegrass and white clover (Di and Cameron, 2002, 2005). In addition, Ca_2^+ and Mg_2^+ leaching were reduced by 51 and 31%, respectively, and herbage DM yield in the urine patch areas was increased by 33% (Di and Cameron, 2005). A review by de Klein and Monaghan (2011) suggested potential reductions in NO_3^- leaching of up to 60% and N_2O emissions by up to 55% with DCD application.

It has also been pointed out that the effectiveness of nitrification inhibitors (specifically DCD) depends largely on temperature, moisture, and soil type. For example, the longevity of DCD decreases with increasing soil temperature (Kelliher et al., 2008; de Klein and Monaghan, 2011). Some studies have suggested potential increases in NH_3 volatilization and NH_4^+ leaching due to increased NH_4^+ accumulation in soil. It has been shown that DCD may not be effective in reducing NO_3^- leaching in soils that leach substantial amounts of NH_4^+, which is also influenced by rainfall (de Klein and Monaghan, 2011).

Results of the combined use of nitrification and urease inhibitors have been inconclusive (Khalil et al., 2009; Zaman and Blennerhassett, 2010). Urease inhibitors inhibit urea hydrolysis to NH_4^+ and thus directly affect substrate availability for NH_3 volatilization. A recent review of the literature using New Zealand as a case study indicated that a urease inhibitor—nBTPT—was effective in inhibiting urea hydrolysis with an average NH_3 emission reduction of 53% ranging from 11 to 93% (Saggar et al., 2012). Because NH_4^+ is also a source of NO_3^- leaching and N_2O emission, it is expected that in-

hibition of urea hydrolysis will affect all 3 pathways of N loss in soil, but this has not been consistently observed (Khalil et al., 2009; Zaman and Blennerhassett, 2010).

As stated earlier, nitrification inhibitors can increase soil NH_4^+ and thus potentially increase NH_3 losses whereas urease inhibitors prolong the stability of urea. If, however, nitrification inhibitor activity is decreased, preservation of N as urea may not decrease consequent losses of N as NO_3^- or N_2O. This scenario is also questioned on the basis of different half-lives of urease and nitrification inhibitors (de Klein and Monaghan, 2011).

Advances in plant biotechnology and microbial enzymology may offer new opportunities for reducing manure-amended soil N_2O emissions. Richardson et al. (2009), based on the idea that soil N_2O emissions from bacterial denitrification processes result from incomplete reduction of N_2O to N_2, suggested potential ways of enhancing this final step in the denitrification process: 1) increasing soil Cu availability to provide sufficient CuA and CuZ, cofactors needed for biosynthesis or assembly of nitrous oxide reductase (**N_2OR**), 2) improving the understanding of the regulation of N_2OR activities (enzyme repair and de novo synthesis), and 3) use of plants to "scrub" N_2O emissions by expressing bacterial N_2OR in plants.

Cover Crops. Cover cropping can reduce soil erosion, improve soil quality and fertility, improve water, weed, disease, and pest management, and enhance plant and wildlife diversity on the farm (Lu et al., 2000; Haramoto and Gallandt, 2004). In some production systems, cover cropping can also increase crop yields (Miguez and Bollero, 2005), reduce input costs, and increase farm profitability by reducing N fertilizer use, improving P availability, and reducing weed control costs (Lu et al., 2000; Stockwell and Bitan, 2012; Kassam et al., 2012).

Reduction of N fertilizer use by growing leguminous cover crops has a direct mitigation effect on soil N_2O emissions by reducing soil NO_3^- availability and potential leaching (Christopher and Lal, 2007). Through their symbiotic relationship with *Rhizobium* (root nodule bacteria), legumes fix atmospheric N, converting it to NH_4^+, which is consequently incorporated into plant AA and proteins. Thus, inclusion of legumes in plant rotation and consequent incorporation of legume residues into agricultural soils enhances plant-available inorganic N and organic soil N (Heichel, 1987). Cover crops can increase plant N uptake and decrease NO_3^- accumulation and thus reduce N_2O production through denitrification, but the results on overall GHG emissions have not been consistent. Interactions with other soil conservation practices exist (tillage system, for example) and must be considered when the goal of cover cropping is to reduce whole-farm GHG emissions. Interactions among soil conservation and management practices, however, are complex and may easily shift the balance of GHG

fluxes. A study from Denmark reported a strong correlation among soil conservation practices, cover cropping, and tillage (Petersen et al., 2011). These authors concluded that reduced tillage may be an N_2O mitigation option in rotations with cover crops but that there was inconclusive evidence that the overall balance of N_2O emissions was positively affected. Another example of these interactions is the study by Garland et al. (2011). These authors demonstrated that differences in cover crop management had the capacity to affect GHG emissions; for example, mowing the cover crop produced larger peak emissions (14.1 g N_2O N/ha per d; no-till system) compared with cover crop incorporation by disking (1.6 g N_2O N/ha per d; conventional tillage system).

A review of soil organic C sequestration and GHG emissions from agricultural activities in the southeastern United States found that combining cover cropping with no tillage enhanced soil organic C sequestration compared with no tillage and no cover cropping (0.53 vs. 0.28 Mg ha/yr; Franzluebbers, 2005). Similar results have been reported for cotton (Causarano et al., 2006), but the C sequestration benefits were minimal and the effect on N_2O emissions were inconsistent in a corn–soybean rotation (Bavin et al., 2009).

Liebig et al. (2010) reported no net GHG mitigation benefit from incorporating a rye cover crop during the fallow phase of a dry land wheat cropping system under no-till management, and similar inconclusive results were reported by a Canadian study (VanderZaag et al., 2011). The simulation analysis of soil conservation practices for several crops (beans, corn, cotton, safflower, sunflower, tomato, and wheat) by De Gryze et al. (2010) in California's Central Valley found that, compared with conventional agricultural management, cover cropping had the largest potential to mitigate soil GHG fluxes resulting in a net reduction of 752 to 2,201 kg CO_2–e/ha per yr (with conservation tillage having the smallest mitigation potential). Similar conclusions were drawn by the authors for alfalfa, melon, and sunflower (De Gryze et al., 2011).

CONCLUSIONS

Conclusions for manure mitigation practices discussed in this review are summarized in Table 1. There are a number of animal and manure management practices that are feasible and can effectively reduce CH_4 and N_2O emissions from manure storage and/or land application. It is important to remember, however, that some of these practices may result in "pollution swapping" or increase NH_3 emissions. Therefore, due to numerous interactions at the animal, storage, and land application phases of the manure management process, GHG mitigation practices should not be evaluated in-

dividually in isolation but as a component of the livestock production system (farm) as a whole.

Optimizing the animal diet to improve N use efficiency, balancing N input with production level, and maintaining fiber digestibility while reducing enteric CH_4 fermentation, are important steps in reducing N_2O and CH_4 emissions from manure. Due to the complex interaction between nutrition, production, animal health, and economic performance, diet modification to reduce N inputs should be done carefully to prevent reduced fiber digestibility and maintain animal productivity.

The type of animal housing used establishes the way feces and urine are handled as well as the storage period of the manure, setting the conditions that would determine the magnitude of GHG emissions and providing opportunities for complementary mitigation practices such as composting, anaerobic digestion, biofiltration, and photocatalytic degradation. Therefore, integral animal housing and manure management system design is an important component in the implementation of GHG mitigation practices.

The choice of manure management technology has a strong influence on energy, nutrient, and GHG balances, and to obtain reliable results, the most representative and up-to-date management technology combined with data representative of the specific area or region must be considered. Overall, lowering the concentration of N in manure, preventing anaerobic conditions, or reducing the concentration of degradable manure C are successful strategies for reducing GHG emissions from manure applied to soil. Semipermeable covers are valuable for reducing NH_3, CH_4, and odor emissions but likely increase N_2O emissions; therefore, their effectiveness is not clear and results may vary widely. Impermeable membranes, such as oil layers and sealed plastic covers, are effective in reducing gaseous emissions but are not very practical.

Biofilters can be an effective tool to reduce CH_4 and NH_3 from mechanically ventilated animal housing facilities, but management of nitrification and dentrification processes in the biofilter are essential to control N_2O emissions and the overall GHG mitigation efficiency of the system. Decreasing manure pH through acidification is a promising practice to reduce NH_3 and CH_4 emissions during manure storage but might increase N_2O emissions following land application. The adoption of manure acidification as a GHG mitigation tool may be limited by the practical and legal regulatory constraints that safe handling of strong acids in a farm setting imposes. Application of acidified manure is not expected to greatly impact crop production; however, long-term impacts of land application of acidified manure on soil pH have not been reported. Composting can be an effective method for reducing GHG emis-

sions from a range of waste materials, including animal manure, but NH$_3$ losses are significant.

Use of anaerobic manure digesters is a GHG mitigation strategy that has a significant potential to capture and destroy most CH$_4$ from manure, generate renewable energy, and provide sanitation opportunities for developing countries. Proper design and management of digestion systems is important so that they do not become net emitters of CH$_4$. There may also be potential for mitigating N$_2$O emissions following land application of the digested manure although results are contradictory. On larger farms, these systems may require large initial capital investments. The adoption of this technology on farms will heavily depend on climatic conditions and the availability of alternative sources of energy. Instruction and technical assistance are also necessary in implementing successful anaerobic digestion mitigation practices. Anaerobic digestion systems are not recommended for geographic locations with average temperatures below 15°C without supplemental heat and temperature control.

Urease inhibitors are effective in reducing urea degradation when applied to animal urine before it is mixed with feces, to drylot pen surfaces (for at least 5 to 10 d), or before urine is deposited in the soil; therefore, excluding manure managements systems that separate feces and urine or continuous treatment of the pen surface, urease inhibitors have limited opportunity to be used effectively in most animal production systems. The use of nitrification inhibitors has been demonstrated as an effective practice to reduce N$_2$O emission from intensive grazing systems but provide minimal if any economical benefit to the producer and therefore it is limited as an attractive mitigation practice.

Manure application techniques such as subsurface injection reduce NH$_3$ and CH$_4$ emissions but can result in increased N$_2$O emissions. Injection works well when combined with anaerobic digestion and solids separation by improving infiltration. Separation of manure solids and anaerobic degradation pretreatments can mitigate CH$_4$ emission from subsurface-applied manure, which may otherwise be higher than from surface-applied manure. Timing of the manure application (e.g., avoiding application before a rain) and maintaining soil pH above 6.5 may decrease N$_2$O emissions. Use of cover crops to reduce the use of commercial fertilizers by increasing N fixation, increase plant N uptake, and reduce the amount of soil N available for NH$_3$ volatilization and N$_2$O production through nitrification is very effective GHG mitigation tool, but when applied to farm systems mixed results have been obtained.

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