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Nitrous Oxide Emissions from Open-Lot Cattle Feedyards: A Review

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Abstract

Nitrous oxide (N_2O) emissions from concentrated animal feeding operations, including cattle feedyards, have become an important research topic. However, there are limitations to current measurement techniques, uncertainty in the magnitude of feedyard N_2O fluxes, and a lack of effective mitigation methods. The objective of this review was to assess N_2O emission from cattle feedyards, including comparison of measured and modeled emission rates, discussion of measurement methods, and evaluation of mitigation options. Published annual per capita flux rates for beef cattle feedyards and open-lot dairies were highly variable and ranged from 0.002 to 4.3 kg N_2O animal^{-1} yr^{-1}. On an area basis, published emission rates ranged from 0 to 41 mg N_2O m^{-2} h^{-1}. From these studies and Intergovernmental Panel on Climate Change emission factors, calculated daily per capita N_2O fluxes averaged 18 ± 10 g N_2O animal^{-1} d^{-1} (range, 0.04–67 g N_2O animal^{-1} d^{-1}). This variation was due to inconsistency in measurement techniques as well as irregularity in N_2O production and emission attributable to management, animal diet, and environmental conditions. Based on this review, it is clear that the magnitude and dynamics of N_2O emissions from open-lot cattle systems are not well understood. Further research is required to quantify feedyard N_2O fluxes and develop cost-effective mitigation methods.

Core Ideas

- The magnitude of published N_2O fluxes from open-lot cattle systems was highly variable.
- N_2O fluxes varied with measurement methods, management, and environmental conditions.
- Methods to mitigate feedyard N_2O emissions are available but have yet to be thoroughly evaluated.
- Improved animal performance and reducing N intake are best options to reduce feedyard N_2O losses.

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Nitrous oxide (N_2O) is a greenhouse gas (GHG) with a 100-yr global warming potential 265 to 298 times greater than carbon dioxide (CO_2) and an atmospheric lifetime of ~114 yr (Myhre et al., 2013; USEPA, 2013). Current tropospheric N_2O concentrations (326 ppb) are lower than that of other GHGs (CO_2, 395 ppm; methane [CH_4], 1893 ppb) (Blasing, 2014); however, small N_2O increases have major impact on climate change due to the ability of N_2O to trap solar energy (i.e., the "greenhouse effect"). In 2007, approximately 60% of global N_2O emissions were attributed to fertilized cropland and livestock production (Smith et al., 2007).

Approximately 51% of anthropogenic N_2O produced in the United States originates from cropland; however, the USEPA estimated that 17.3 Tg CO_2 equivalents (CO_2eq) of N_2O were derived from livestock manure in 2015, of which 8 Tg CO_2eq were attributed to beef cattle (USEPA, 2015). The estimated contributions of major livestock sectors to manure-derived N_2O in the United States are presented in Fig. 1a, where the impact of beef and dairy industries is clear.

In 2009, the USEPA proposed a mandatory reporting rule for livestock facilities that annually generate 25,000 t of CO_2eq as CH_4 and/or N_2O from manure (USEPA, 2009). The estimated threshold cattle population for generation of such a large quantity of GHG at a single beef operation was 88,923 head. Approximately 80 to 90% of beef cattle in the United States are finished in open-lot feedyards with holding capacities of more than 1000 animals: in 2007 a total of 774 feedyards had capacities greater than 2500 cattle (USDA–NASS, 2007). Many of the top 20 beef producing companies in the United States have

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Abbreviations: AOA, ammonia-oxidizing archaea; AOB, ammonia-oxidizing bacteria; CAFO, concentrated animal feeding operation; CP, crude protein; CTC, chlortetracycline; DCD, dicyandiamide; DEA, denitrification enzyme activity; DMPP, 3,4-dimethylpyrazole phosphate; DNDC, Denitrification and Decomposition Model; EF, emission factor; EC, electrical conductivity; GC, gas chromatograph; HIP, "hole-in-the-pipe"; IFSM, Integrated Farm Systems Model; IPCC, Intergovernmental Panel on Climate Change; MM, micrometerological method; NA, nitrification activity; NBPT, *N*-(*n*-butyl) thiophosphoric triamide; NFT-NSS, non–flow through/non–steady state; NOB, nitrate-oxidizing bacteria; OM, organic matter; WDGS, wet distillers grain plus solubles.

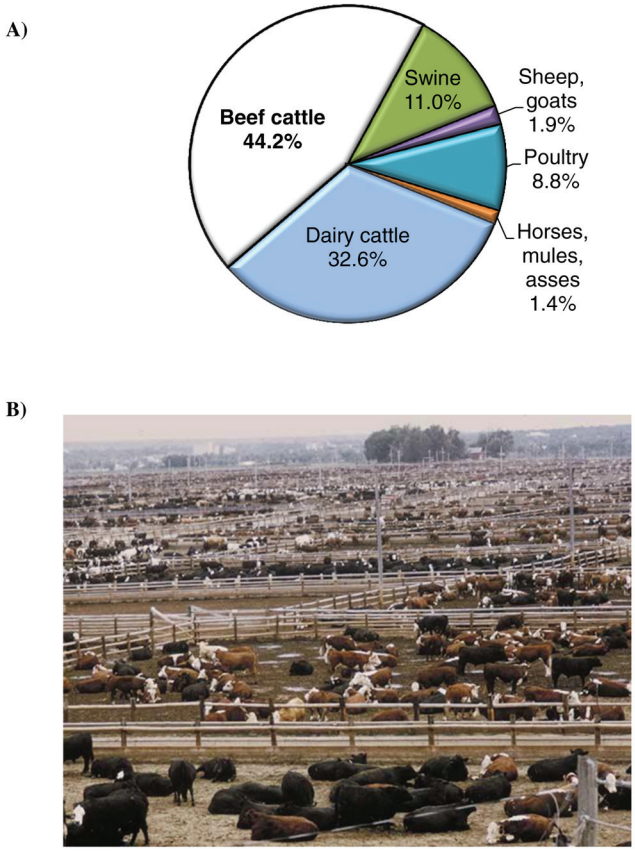

Fig. 1. (A) Estimated contributions of major livestock sectors to manure-derived N_2O in the United States. Data adapted from USEPA (2013). (B) Commercial beef cattle feedyard in the Texas Panhandle. Finishing cattle are typically housed in earthen-surfaced pens for approximately 180 d at a density of approximately 15 m^2 $animal^{-1}$.

feedyards with capacities >100,000 cattle (NCBA, 2015). Most large feedyards are located in the southern High Plains of Texas, Nebraska, and Kansas (USDA–ERS, 2012). The number of large feedyards and open-lot dairies is continually expanding in the southern High Plains (USDA–NASS, 2015). An example of a large beef feedyard in Texas is presented in Fig. 1b.

Regulating N_2O Emissions from Cattle Feedyards

Approximately 23 million cattle are fed annually in US feedyards with capacities >1000 animals (USDA–NASS, 2013). Most are housed in open-lot, soil-surfaced pens. The manure (a mixture of feces, urine, soil, dropped feed, and scurf) that accumulates in pens is heterogeneous and dynamic, consisting of both freshly excreted and older material that is continually changing compositionally. Manure is a complex mixture of H_2O, organic matter (OM), nutrients, microbes, enzymes, minerals, and pharmaceutical residues. Manure undergoes numerous (bio)chemical processes, including mineralization, hydrolysis, nitrification, denitrification, and fermentation. To date, it is assumed that most feedyard N_2O is produced via nitrification and denitrification, although the complex nature of the manure pack itself, the high spatial variability of manure characteristics within pens, and interactions among environmental and management factors complicate this interpretation (Cole et al., 2009a,b; Waldrip et al., 2015a,b,c,d).

Environmental issues occur when cattle are housed in high densities, largely due to manure accumulation within a relatively small area. In general, less than 20% of the nitrogen (N) consumed by cattle is converted into animal product, with the remainder excreted in urine and feces (Cole and Todd, 2009; Mosier et al., 1998). Imbalances between feedyard N imports (e.g., purchased feed and animals) and exports (e.g., animal product and manure) can cause N losses to air and water. This review focuses on N_2O emissions; however, Waldrip et al. (2015a,b) recently conducted detailed reviews on general feedyard N sustainability and ammonia (NH_3) emission.

In the United States, concentrated animal feeding operations (CAFOs) have been subjected to regulation, reflecting heightened concern over pollution from livestock. Manure is estimated to be the source of ~17% of N entering US waters (Smith et al., 1997). Clean Water Act (33 U.S.C. §1251 et sEq. [1972]; USEPA, 2002) regulations require that CAFOs with a National Pollutant Discharge Elimination System permit develop and implement management plans for fields that receive manure (USEPA, 1990). Such a plan, which must meet USDA Natural Resources Conservation Service (NRCS) standards, limits nutrient application to land and specifies control measures to prevent nutrient losses. However, implementing and enforcing feedyard adoption of nutrient management plans may be cost prohibitive. The USDA Economic Research Service estimates that reductions in net returns in the livestock and poultry sectors due to regulation would be about $1.4 billion per year, with subsequent economic welfare decline for producers and consumers by almost $2 billion per year (Ribaudo et al., 2003). A benefit of regulation would be improved air, but at a cost to both producers and consumers.

Organic and inorganic N in manure is converted to forms that are readily transported by hydrologic and atmospheric processes. Focusing strictly on one issue, such as nitrate (NO_3^-) leaching, could increase emissions of N_2O, NH_3, or other gaseous N compounds involved in environmental degradation. As an example, a market for N_2O offsets may reduce total N emission to air but increase N losses to water. Pollution swapping tradeoffs complicate development of best management practices to prevent N losses.

Quantification of feedyard N_2O emission is needed for inventory purposes and to estimate the impact of cattle production on the environment. Programs, such as NRCS's Conservation Innovation Grants (USDA–NRCS, 2002), may provide funding for CAFO to initiate measures to mitigate N losses; however, it is unclear to what extent feedyard operators have sought funding for N_2O mitigation.

N_2O from Beef Cattle Feedyards

Nitrous oxide losses from open-lot feedyards are typically very small compared with emissions of NH_3 and N_2O from agricultural soils and NH_3 emission from animal housing. More compelling aspects of controlling and understanding feedyard N_2O are increased atmospheric GHG concentrations and potential CAFO reporting and regulation standards. Beef cattle researchers and producers have traditionally focused on increasing production efficiency, with less emphasis on long-term consequences; this approach may need revising to preserve economic viability of the beef industry while maintaining the environment.

N_2O Formation Processes

A substantial body of work exists on N_2O formation and emission from soils, which was reviewed by Butterbach-Bahl et al. (2013). Less is known about manure-derived N_2O, although some studies have analyzed N_2O emission and related processes during manure composting or after soil application (Fortuna et al., 2012; Maeda et al., 2010). Various biotic and abiotic mechanisms produce N_2O from soils, including denitrification, nitrifier-denitrification, nitrification, coupled nitrification–denitrification, dissimilatory nitrate ammonification, chemical decomposition, and chemodenitrification (Butterbach-Bahl et al., 2013). It is assumed that manure-derived N_2O is primarily produced through microbial nitrification and denitrification (Jungbluth et al., 2001; Maeda et al., 2010). There is increasing evidence that abiotic N_2O production via hydroxylamine (NH_2OH) oxidation and chemodenitrification is important in soils (Zhu-Barker et al., 2015). To our knowledge, no research has assessed the role of abiotic processes in manure-derived N_2O emission.

Nitrification

Nitrification is the aerobic oxidation of reduced N (usually ammonium [NH_4^+] or NH_3) by ammonia-oxidizing bacteria (AOB) and ammonia oxidizing archaea (AOA) to nitrite (NO_2^-) (Fig. 2). Nitrite is then oxidized by nitrate-oxidizing bacteria (NOB) to NO_3^-. Fungi and other heterotrophic nitrifiers also produce NO_3^- from organic material (Laughlin et al., 2009). In these processes, nitric oxide (NO) and N_2O are gaseous intermediates that may be released from incomplete reactions. The formation of NO_2^- requires two enzymes: ammonia monooxygenase (AMO; *amoA*) and hydroxylamine oxidoreductase (Norton, 2008). Reduction of NO_2^- to NO_3^- is performed by nitrite oxidoreductase. The initial oxidation of NH_4^+ to NO_2^- is usually the rate-limiting step for nitrification (Gruber and Galloway, 2008); therefore, chemical compounds that block this reaction are one method to reduce N_2O emissions (Dalal et al., 2003).

Current understanding of manure nitrification is largely derived from agricultural soils. This notion may be invalid due to soil and manure differences in OM and N contents and physicochemical properties. Nitrification potential, an estimate of the maximum capacity of microbes to transform NH_4^+ into NO_3^-, has been determined on manure-amended soils (Fortuna et al., 2012; Laanbroek and Gerards, 1991). The quantitative polymerase chain reaction technique has elucidated the nitrifier community structure in manure, allowing for direct assessment of *amoA* and AOA- and AOB-specific 16S rRNA gene copies. Yamada et al. (2013) found that AOA were the predominant nitrifiers during the early stages of beef manure composting. Rice et al. (2007) characterized the microbial community of beef feedyard manure, where both community structure and population numbers of AOB and NOB differed both with season and depth within the manure pack.

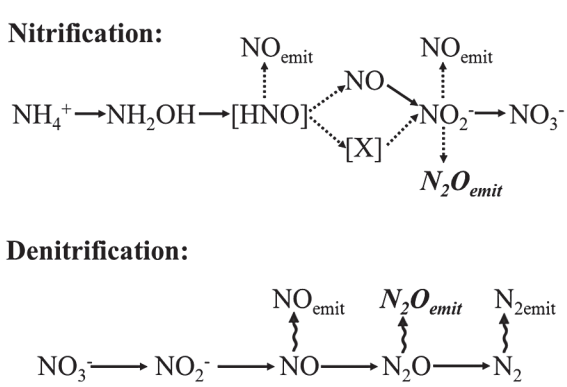

Fig. 2. Pathways of nitrification and denitrification leading to N_2O emission from soils and manures. Dashed lines and square brackets indicate incompletely understood processes and intermediates, respectively.

In soils, nitrification is controlled by oxygen and H_2O content, temperature, and NH_4^+/NH_3 availability (Norton, 2008). Recent research indicates that the same may be true for beef manure (Ayadi et al., 2015). The optimum H_2O content for soil nitrifiers is about 60% of available pore space (Brady and Weil, 2002), but studies on H_2O content effect on manure nitrification rates are sparse. Waldrip et al. (2015d) saw that N_2O emission from feedyard manure at 20% saturation was 41% higher than at 80% saturation, suggesting enhanced nitrifier activity when manure was dry. In contrast, Miller and Berry (2005) reported negligible N_2O emission from incubated soil/manure mixtures (25% soil and 75% beef cattle manure), despite variations in H_2O content. Further studies are warranted to determine the effect of H_2O content on N_2O emissions from feedyards. Soil nitrification rates increase with temperature, with an optimum between 27 and 35°C, but are reduced at temperatures <10°C (Sabey et al., 1956). Similarly, Ayadi et al. (2015) observed that potential nitrification activity in packed beef manure was 120% higher at 40°C than at 10°C.

Soil nitrifier community structure and nitrification rates are influenced by N availability (Fortuna et al., 2012). For example, AOB populations tended to increase with increased NH_4^+ concentrations in N-rich grassland soils (Di et al., 2010), whereas AOA were more predominant relative to AOB under N-limited conditions (Hatzenpichler, 2012). Specific activities of AOB are generally greater than AOA; however, AOA activity can be higher under N-limited conditions (Herrmann et al., 2008; Taylor et al., 2010). Fortuna et al. (2012) used quantitative polymerase chain reaction on a diverse soil set amended with dairy slurry, where AOB copy numbers were an order of magnitude greater than AOA. There was a tendency for AOB, but not AOA, to increase with slurry application; however, results were inconsistent among the soils studied. Overall, soil nitrification potential was related to AOB copy numbers and total organic C and total N concentrations.

Little is known about the fungal contribution to nitrification in manure. Rice et al. (2007) saw low fungal diversity in the wet-pack, the dense layer of accumulated pen manure that has a higher H_2O content than overlying manure (Cole et al., 2009b), and proposed that fungal nitrification could be limited by bacterial competition for substrate and low O_2 concentrations. Maeda et al. (2011) pinpointed a need for research on fungal N_2O production in composts. Antibiotics are commonly used at subtherapeutic levels in commercial feedyards to promote weight gain and improve animal health. Chlortetracycline (CTC), a broad-spectrum antibiotic, is active against a range of gram-positive and gram-negative bacteria and protozoa and is administered to cattle for disease prevention; however, approximately 75% of CTC administered to cattle may be excreted (Elmund et al., 1971). Monensin sodium (Rumensin, Elanco Animal Health) is an ionophore used to control coccidiosis and rumen acidosis and

to improve beef cattle feed efficiency. According to the manufacturer, estimated monensin sodium excretion rates were 9.5 mg kg^{-1} manure when supplied to finishing cattle at high levels (480 mg $animal^{-1}$ d^{-1}) (Elanco, 2005). Tylosin phosphate (Tylan, Elanco Animal Health) is a macrolide antibiotic used to treat pneumonia and foot rot, promote growth, prevent liver abscesses, and control other common cattle diseases. Manure from beef cattle that received 11 mg d^{-1} tylosin phosphate contained concentrations of 107 mg tylosin phosphate kg^{-1} manure (Amarakoon et al., 2016).

Antibiotic residues may reduce the contribution of bacterial-derived N_2O, causing fungi to predominate N_2O production on feedyards. Hao et al. (2011) measured N_2O emissions from composting manure where cattle were fed diets containing CTC, CTC plus sulfamethazine, or tylosin phosphate. These researchers noted that N_2O from compost made from manure from cattle treated with CTC/sulfamethazine, CTC, and tylosin phosphate tended to be greater than that from untreated cattle, but the results were inconsistent for the 2 yr of the study period. To date, no comprehensive study has assessed the role of fungi on feedyard N_2O emissions.

In contrast to soils, nitrification may not be a predominant process leading to N_2O emissions from feedyard manure because some common nitrifiers are sensitive to high NH_4^+/NH_3 concentrations. Suwa et al. (1994) found that NH_3 oxidizers in activated sewage sludge were inhibited when NH_4^+ concentrations were greater than 5 mmol L^{-1} (90 mg NH_4^+ L^{-1}). Koper et al. (2010) reported nitrification inhibition in soils amended with dairy manure compost. In contrast, Antileo et al. (2002) observed nitrifyer acclimatization to NH_3–N concentrations as high as 13 mg L^{-1} in sewage sludge. Typical NH_3/NH_4^+–N concentrations in Texas feedyard pens averaged approximately 6755 mg kg^{-1} in fresh urine spots and 2281 mg kg^{-1} in nonurine spots (Cole et al., 2009a,b). It is unknown how these high NH_3/NH_4^+ concentrations affect nitrification in feedyards.

Denitrification

Denitrification generally occurs under anaerobic and microaerophilic conditions, where NO_3^- is reduced to N_2 by a variety of microbial groups, including heterotrophic and autotrophic bacteria, Archea, Proteobacteria, and eukaryotic fungi (Zumft, 1997). The denitrification pathway involves the sequential action of dissimilatory NO_3^- reductase, dissimilatory NO_2^- reductase, NO reductase, and N_2O reductase (Fig. 2). Although manure denitrification has been studied, as with nitrification, most understanding comes from the considerable body of soil denitrification research. In both soils and manures, denitrification rates are largely controlled by temperature, pH, H_2O, and NO_3^- contents, and N_2O reductase activity (Galbally, 1989; Tsutsui et al., 2013; Willers et al., 1993).

Bremner and Shaw (1958) reported that soil denitrification rates increased with temperatures from 2 to 25°C. Optimal soil denitrification temperatures range from 25 to 45°C (Brady and Weil, 2002; Lensi and Chalamet, 1982). Woodbury et al. (2001) measured denitrification enzyme activity (DEA) and nitrification activity (NA) in feedyard manure collected from different depths (loose surface manure, manure pack at depths of 0–0.1 and 0.10–0.20 m) and locations within feedyard pens (near feedbunk, manure mounds, and near downgradient fenceline). The highest DEA rates were in loose surface manure near the downgradient fenceline (132 mmol L^{-1} g^{-1} h^{-1}), whereas NA was greatest in the 0- to 0.1-m depth of the manure pack at the base of manure mounds (462 mmol L^{-1} g^{-1} h^{-1}). Thus, spatial variability in pen manure plays a role in determining the predominant mechanism producing feedyard N_2O.

The catabolism of fecal carbohydrates produces residual fatty acids that create anaerobic conditions when oxidized (Paul and Beauchamp, 1989). Nielsen et al. (1996) reported that high microbial respiration rates also cause anaerobic conditions in manure. High bulk density (~1.7–1.9 g cm^{-3}) and low air and H_2O permeability (~1.9×10^{-6} to 6.2×10^{-7} cm s^{-1}) contribute to anoxic or anaerobic conditions favorable for denitrification in feedyard pens (McCullough et al., 2001; Mielke and Mazurak, 1976). However, these same conditions can inhibit diffusion of NH_3 and N_2O from the manure pack to the atmosphere (Williams et al., 2008).

Substrate pH influences whether N_2O or N_2 is the final denitrification product. In soil, N_2O is the major product at pH <7, whereas N_2 predominates at pH >8 (Simek and Cooper, 2002; Wijler and Delwiche, 1954). Feedyard manure pH usually ranges between 7.1 and 8.4 and increases with urine application (Cole et al., 2009a,b; Ward et al., 1978); thus, N_2 should be the primary denitrification product. However, pH influence on denitrification is complicated by high salt concentrations (electrical conductivity [EC], ~1.0 vs. 0.3 dS m^{-1} for soil underlying feedyard pens) (Cole et al., 2009a), which can inhibit denitrification (Woodbury et al., 2001). Further study is needed to elucidate the interactions between pH and EC on feedyard N transformations.

Interaction between Nitrification and Denitrification in N_2O Production

The particular pathway that produces N_2O in soils is largely controlled by H_2O content, where nitrification predominates when water-filled pore space is <60% (Pihlatie et al., 2004). Pihlatie et al. (2004) reported that nitrification in peat soils accounted for 76, 42, and 22% of total N_2O production at 60, 80, and 100% water-filled pore space, respectively. Denitrification rates increase exponentially with soil H_2O content >55% and are highest near saturation (Parton et al., 2001). The same trend appears to be true for manure because nitrification was the source of N_2O from dairy manure at high redox potential (low water content) and denitrification predominated at low redox potential (Brown et al., 2000).

Evidence links nitrification and denitrification to total N_2O production in soil (Nielsen et al., 1996). Firestone and Davidson (1989) conceptualized the "hole-in-the-pipe" (HIP) soil model that relates production of NO and N_2O to N availability, where the specific production mechanism is a function of H_2O content. In the HIP model, N_2O escapes from the theoretical holes in the pipe: the size of the holes and extent of N_2O emission is primarily based on soil H_2O content. If HIP is valid for feedyard manure, high spatial and temporal variability in pen manure H_2O and N content complicates determination of the primary N_2O production mechanism and complicates development of effective mitigation strategies to control feedyard N_2O emission.

Increasing manure H_2O content could slow nitrification, accelerate denitrification, or result in a combination of these processes. Petersen et al. (2004) and Monaghan and Barraclough (1992) noted little accumulation of NO_3^- in pasture soil during

the first 2 to 5 d after urine application. Limmer and Steele (1982) saw increased soil NO_3^- concentrations 6 to 11 d after urine application, with decreased denitrification potential after 21 d. In contrast, Petersen et al. (2004) noted increased soil NO_3^- content that accounted for 17 to 23% of applied N just 4 d after urine application. Monaghan and Barraclough (1992) hypothesized that the combined inhibitory effects of elevated pH, high NH_3 concentrations, and osmotic stress reduced nitrification rates in pasture urine spots. Similar conditions exist in feedyard pens; however, high stocking densities result in more frequent urine application within a limited area than in grazing systems.

One example of H_2O content effects on feedyard N_2O emission comes from Aguilar et al. (2011), who used static flux chambers on a commercial Kansas feedyard and noted that flux rates and magnitude of N_2O emissions varied greatly depending on pen conditions. The greatest (25 mg N_2O m^{-2} h^{-1}) and most variable emission rates were from moist/muddy spots, as compared with dry spots or flooded areas. Moist/muddy spots are considered "hot spots" (localized micro-sites with conditions favoring microbial activity) for N_2O production and emission (Matthews et al., 2010; Woodbury et al., 2001). Excessive H_2O content (e.g., flooded conditions) reduces redox potential and decreases air-filled pore space, which prevents gas diffusion and subsequent emissions. Temperature is an important factor in N_2O production due to its influence on OM decomposition and N mineralization (Saggar et al., 1998). Woodbury et al. (2001) proposed that the loose, well-aerated surface manure in feedyard pens was highly organic and favorable for both organic N mineralization and nitrification, whereas compacted manure at lower depths contained anaerobic zones promoting denitrification. Thus, feedyard N_2O is likely a result of combined denitrification and nitrification processes, similar to observations from compost and soil (Ma et al., 2008; Maeda et al., 2010).

Enteric N_2O Production

In addition to manure-derived N_2O, some enteric N_2O may be produced by ruminant animals (Kaspar and Tiedje, 1981; Petersen et al., 2015; Stackhouse et al., 2011). The contribution of enteric N_2O to the environmental footprint of cattle has not been firmly established. Kaspar and Tiedje (1981) suggested that ruminal microbes produce small amounts of N_2O as a byproduct of dissimilatory NO_3^- reduction to NH_4^+. In contrast, Reynolds et al. (2010) reported no enteric N_2O emissions from cattle in respiration chambers.

To date, enteric N_2O has been largely ignored in models and GHG inventory calculations. Petersen et al. (2015) saw enteric N_2O emissions ranging from 0.4 to 67 g CO_2eq kg^{-1} dry matter (DM) intake from dairy cows in respiration chambers. Stackhouse et al. (2011) reported that beef calves and steers emitted from 0.7 to 19.9 mg N_2O h^{-1} in respiration chambers, with higher values emitted by larger animals. In a study to evaluate the effects of supplemental dietary NO_3^- on CH_4 emissions from dairy cows, N_2O ranged from 0.4 to 67.2 g CO_2eq kg^{-1} DM intake (Petersen et al., 2015). Supplementation at 21 g NO_3^- kg^{-1} DM decreased CH_4 emission by 19 to 47%, but N_2O increased by as much as 66 g CO_2eq kg^{-1} DM intake. More research is needed to quantify and evaluate diet effects on enteric N_2O production.

Quantifying Feedyard N_2O Emissions

Emissions of N_2O from open-lot cattle systems are not continuous and are highly variable (Table 1). Heterogeneity in measured N_2O fluxes is due to interactions among the chemical, physical, and biological properties that control nitrification and denitrification. In addition, the specific method used to measure and calculate N_2O flux can lead to variability in reported per capita N_2O emissions for cattle, which complicates comparison among published studies. Calculated daily per capita N_2O emissions measured with different methods, emission factors (EFs), and model predictions are presented in Fig. 3, where the differences in N_2O estimates among studies and methods used are clear. Data from Intergovernmental Panel on Climate Change (IPCC) EFs were converted to daily N_2O production based on average N excretion of cattle fed crude protein (CP) at the recommended level of 12.5 to 13.5% dry matter intake (NRC, 2000) or fed higher CP diets containing wet distillers grain plus solubles (WDGS). Data presented on an area basis (Aguilar et al., 2014; Boadi et al., 2004; Redding et al., 2015) were assumed to house finishing cattle at a density of 15 m^2 per animal.

N_2O Measurement Methods

Most frequently, N_2O from livestock operations is measured with nonflow through/nonsteady state (NFT-NSS) chambers where N_2O concentrations are measured with a gas chromatograph (GC) equipped with an electron capture detector (Cortus et al., 2015; Costa et al., 2014). An example of NFT-NSS chambers installed in a feedyard pen is presented in Fig. 4a. The NFT-NSS chambers were used to measure N_2O production from simulated manure packs by Ayadi et al. (2015), where the effects of temperature were demonstrated after application of fresh feces and urine (Fig. 5).

Using the NFT-NSS method, a chamber (generally polyvinyl chloride or some other inert material) with a known volume is inserted into the manure and capped for short periods (generally 30 min) to collect emitted compounds. Unlike active chambers and wind tunnels where sweep air is passed through the chamber at a known rate, no sweep air is passed through NFT-NSS; rather, N_2O diffuses from the manure into the chamber headspace. Headspace N_2O concentration increases over time, and air samples are withdrawn periodically and analyzed with a GC–electron capture detector. Flux rates are calculated from the change in N_2O concentration over time, usually according to the USDA Agricultural Research Service GRACEnet (Greenhouse gas Reduction through Agricultural Carbon Enhancement) protocol developed for soils (Hutchinson and Mosier, 1981; Parkin and Venterea, 2010), although the specific calculation method may vary.

Typical issues with NFT-NSS chambers are lateral diffusion and leakage when the surface is irregular (e.g., surface roughness), high spatial variation where coefficients of variation commonly exceed 100% (Parkin and Venterea, 2010), difficulty inserting chambers to optimal depths (Hutchinson and Livingston, 2001, 2002), and creation of micro-environments within the chamber (e.g., temperature, humidity and pressure perturbations, gas mixing, and manure disturbance). Chamber flux measurements are usually determined on hourly to daily scales because of labor-intensive sample collection and the need for numerous chambers to account for spatial heterogeneity. Thus, short-term N_2O fluxes, such as from urine spots, may not be captured with this

Table 1. Selected studies of N2O emissions from open-lot cattle production systems. All data are presented in units reported in original literature. A graphical comparison of calculated annual per capita N2O emissions is presented in Fig. 3.

Reference	Cattle production system	N_2O emission rate	Technique
IPCC (2006)	emission factor developed for dry lot and bedded cattle	0.02 kg N_2O–N kg^{-1} N excreted (dry lot); 0.01 kg N_2O–N kg^{-1} N excreted (deep bedded manure, no mixing); 0.07 kg N_2O–N kg^{-1} N excreted (deep bedded manure, active mixing)	average values derived from literature review
Phetteplace et al. (2001)	simulated US feedyard	4.3 kg N_2O $animal^{-1}$ yr^{-1}	modeled with IPCC method
Boadi et al. (2004)	Manitoba, Canada research feedyard	0.06 kg N_2O $animal^{-1}$ yr^{-1}	vented small chambers (20.3 cm diameter, 0.03 m^2 footprint)
Borhan et al. (2011)	Texas panhandle feedyard	0.21 kg N_2O $animal^{-1}$ yr^{-1}; 3.4 g animal d^{-1} (summer)	dynamic chamber (0.19 m^2 footprint) and portable gas chromatograph
Leytem et al. (2011)	southern Idaho open-lot dairy	3.7 kg N_2O $animal^{-1}$ yr^{-1} (0.2–33 g N_2O $animal^{-1}$ d^{-1})	photoacoustic field gas monitor and inverse dispersion method
Redding et al. (2015)	beef feedyards in Northern and Southern Australia	0.35 kg N_2O $animal^{-1}$ yr^{-1} (Southern Australia); 0.002 kg N_2O $animal^{-1}$ yr^{-1} (Northern Australia)	large vented nonflow-through chamber (20 m^2 footprint)
Rahman et al. (2013)	North Dakota feedyard	0.03 g N_2O $animal^{-1}$ d^{-1}	portable wind tunnel (0.80 × 0.40 m) with airflow rate of 1.75 m^3 s^{-1}
Costa et al. (2014)	Brazilian feedyard	0.02 g ± 27.1 g N_2O–N $animal^{-1}$ d^{-1}	static chambers
Bonfacio et al. (2015)	simulated Idaho open-lot dairy	14 g N_2O $animal^{-1}$ d^{-1}	modeled with the Integrated Farm Systems Model
Cortus et al. (2015)	monoslope beef barns	0.89–1.0 g N_2O $animal^{-1}$ d^{-1}	gas filter correlation analyzer
Stackhouse-Lawson et al. (2013)	California research feedyard	0.24–0.29 g N_2O $animal^{-1}$ h^{-1} (~6.4 g N_2O $animal^{-1}$ d^{-1})	enclosed 185 m^2 cattle pens vented to a photoacoustic field gas monitor.
Zhu et al. (2014)	milking cows and heifers on an open-lot dairy in China	1.5–2.8 g $animal^{-1}$ h^{-1} (~ 37 g N_2O $animal^{-1}$ d^{-1}; milking cows); 0.3–2.2 g $animal^{-1}$ h^{-1} (~ 24 g N_2O $animal^{-1}$ d^{-1}; heifers).	gas samples collected from nine inlet ports along 80 m length of tubing into a mixing chamber and analyzed with gas chromatography.
Boadi et al. (2004)	Manitoba, Canada feedyard	0.134 mg N_2O m^{-2} h^{-1}	vented small static chambers (20.3 cm diameter, 0.03 m^2 footprint)
Aguilar et al. (2014)	Kansas feedyard under differing pen conditions	0.0–41.4 mg N_2O m^{-2} h^{-1} (median values: moist/muddy, 2.0; dry/loose, 0.16; dry/compacted, 0.13; flooded, 0.10 mg N_2O m^{-2} h^{-1})	static chambers
Redding et al. (2015)	beef feedyards in Northern and Southern Australia	0.004 kg N_2O ha^{-1} d^{-1} (Southern Australia); 0.43 kg N_2O ha^{-1} d^{-1} (Northern Australia)	large vented nonflow-through chamber (20 m^2 footprint)

method. However, NFT-NSS chambers are suitable for evaluating mitigation methods and determining the effects of changing manure physicochemical characteristics and environmental conditions on N_2O fluxes (Hensen et al., 2013).

Selection of an appropriate mathematical method to relate measured N_2O concentrations to flux is essential for NFT-NSS data. Different methods can produce quite disparate results, creating considerable uncertainty in N_2O flux estimates (Levy et al., 2011; Venterea, 2013). Some calculation methods include linear or quadratic regression (Hutchinson and Mosier, 1981), the nonlinear diffusive flux estimator method (Livingston et al., 2006), the chamber bias correction method (Venterea, 2010), and the revised Hutchinson and Mosier method (Pedersen et al., 2010). Parkin and Venterea (2010) developed a comprehensive manual for using NFT-NSS chambers for soil emissions, but it is unclear to what degree these guidelines are used for CAFO N_2O studies. Standardized protocols with NFT-NSS chambers are needed that include strategies for typical complications in feedyards (e.g., difficulty inserting chambers into the dense manure pack, surface irregularity, spatial variability, and inability to leave chambers in pens occupied by cattle because of potential damage to equipment by animals or animal injury). Many of these same concerns have been an issue for quantification of NH_3 and other emissions from cattle pens with chambers and wind tunnels (Cole et al., 2007; Parker et al., 2009, 2013).

Micrometerological methods (MMs) rely on measured atmospheric gas concentrations and a description of atmospheric processes (e.g., wind speed and direction, air temperature, turbulence) near the emitting surface. Advantages of MMs include noninterference with emission processes, integration over large areas (e.g., m^2 or ha), and measurements on time scales <1 h. Some disadvantages of MMs include the requisite specialized equipment, technical expertise, and relatively large source areas. Calculation of trace gas emission rates using MMs may be complicated when the emission source is mobile, such as grazing or feedyard cattle. Feedyard N_2O concentrations are often barely detectable above background; thus, current MMs may not capture small and episodic N_2O fluxes from urine spots or areas with high DEA or NA.

Open-path Fourier-transform infrared spectroscopy, tunable diode laser spectroscopy, eddy covariance, and other MMs have been used to measure NH_3, N_2O, and CH_4 emissions from open-lot cattle systems (Baum, 2008; Baum and Ham, 2009; Leytem et al., 2011, 2013; Todd et al., 2011). Singurindy et al. (2007) used eddy covariance to determine N_2O emission from soil after dairy manure application. Eddy covariance systems, such as pictured in Fig. 4b, have potential as a sensitive method to quantify N_2O fluxes from open-lot systems but have yet to be deployed in feedyards.

Using a novel approach, Zhu et al. (2014) collected air samples through nine inlet ports along an 80-m length of polyethylene tubing at an open-lot dairy in China. Line-averaged N_2O concentrations were determined with a GC, and flux rates were calculated with the backward Lagrangian stochastic inverse dispersion technique (Flesch et al., 2009).

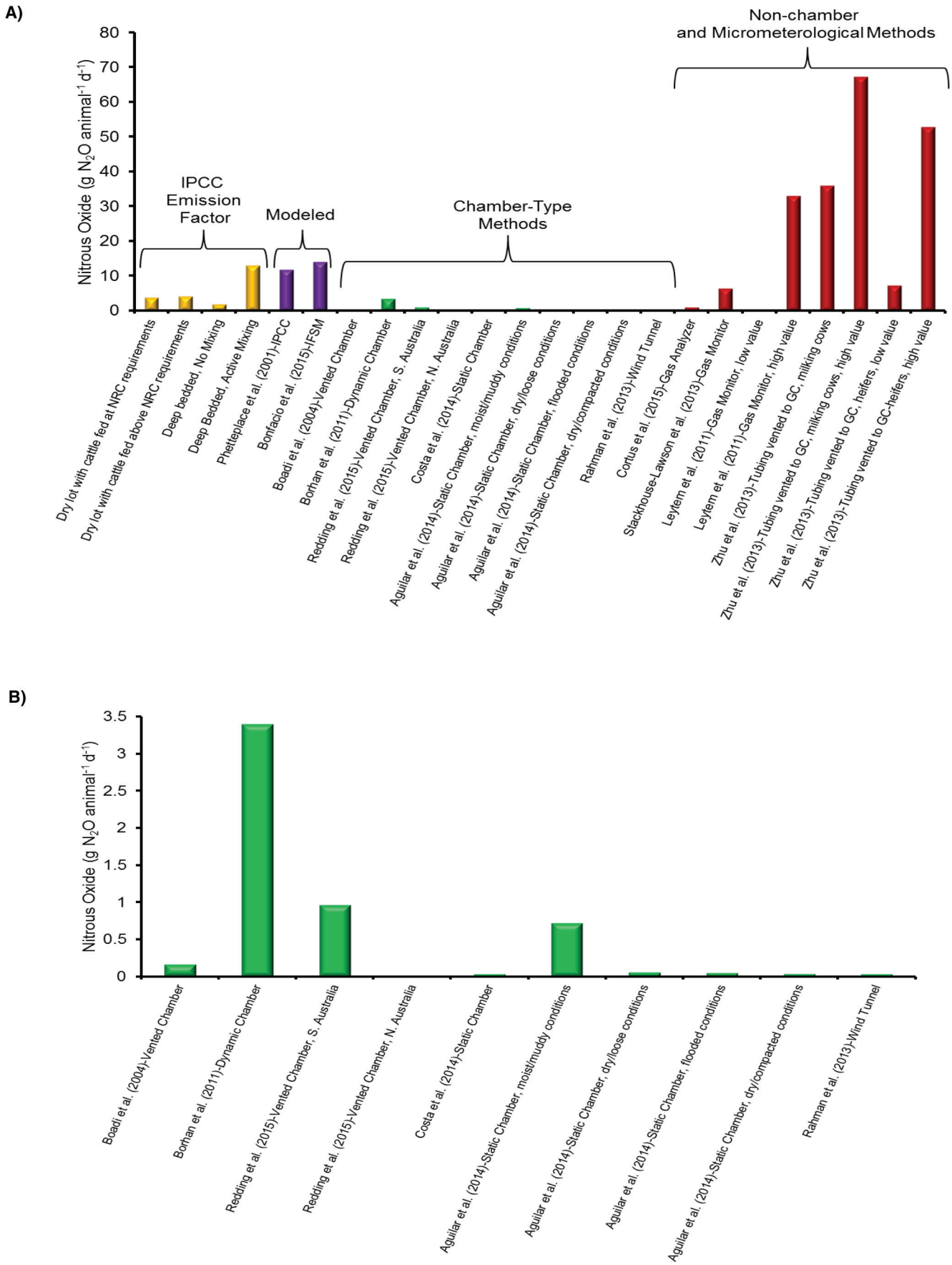

Fig. 3. Comparison of Intergovernmental Panel on Climate Change (IPCC) emission factors, model predictions, and measurement methods on annual per capita N_2O emission from open-lot cattle production systems. All data were converted from the original units presented in Table 1 to kg N_2O animal^{-1} yr^{-1}. GC, gas chromatography; NRC, National Research Council.

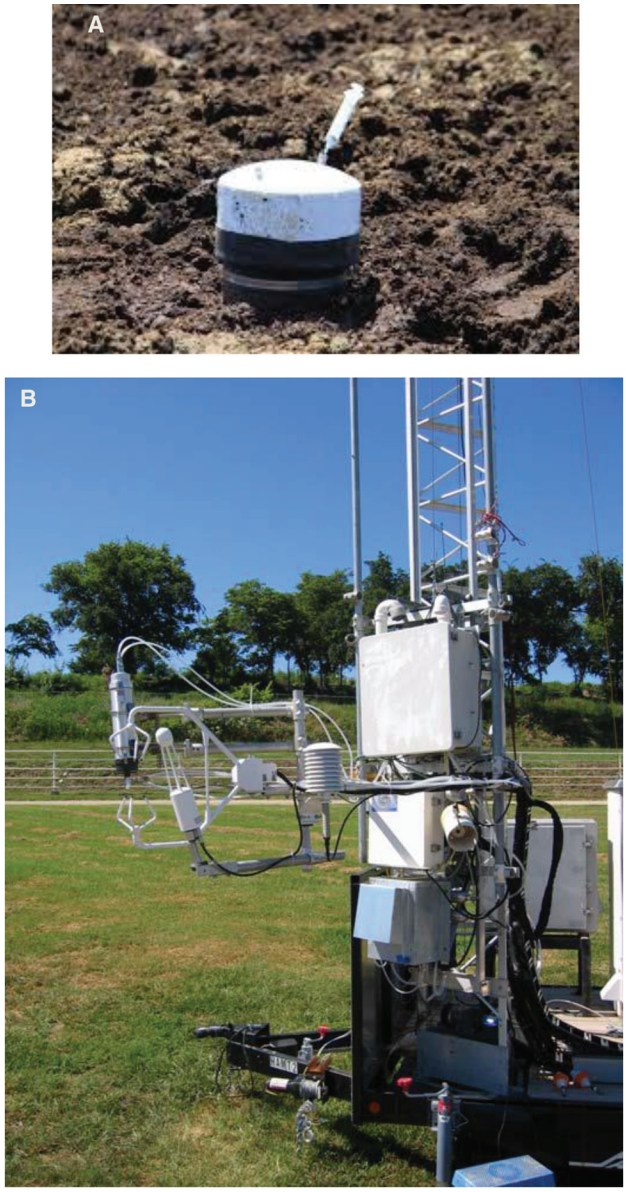

Fig. 4. Examples of available methods to measure N_2O and ammonia in livestock production. (A) Sealed nonflow-through/nonsteady state chamber deployed in a feedyard pen. (B) Relaxed eddy covariance system used by Baum (2008).

Emission rates for milking cows ranged from 1.5 to 2.8 g N_2O animal^{-1} h^{-1}, whereas emission from heifers were 0.3 to 2.2 g N_2O animal^{-1} h^{-1} (Fig. 6) (Zhu et al., 2014). Differences between N_2O fluxes in milking cows and heifers were attributed to manure management, with weekly manure removal from milking cow housing versus twice monthly removal for heifers. Another factor could be higher urea-N excretion by milking cows than heifers due to greater feed demand and N intake by lactating animals (Cassel et al., 2005). Higher temperatures were recorded in milking cow areas (~27°C) than that for heifers (~12°C) by Zhu et al. (2014), which could lead to greater N_2O production and emission.

Scale of Feedyard N_2O Emissions

Limited research has been conducted to directly measure N_2O emission from cattle feedyards; thus, the impact of N_2O from open-lot cattle systems on global climate change remains unclear. Some work suggests that open-lot systems may not be a significant N_2O source. Stackhouse-Lawson et al. (2013) used a photoacoustic infrared field gas monitor to measure N_2O emission from steers housed in four completely enclosed pens (40 steers per each 185-m^2 soil-surfaced pen). Per capita N_2O emissions were ~0.24 g N_2O animal^{-1} h^{-1}, or 6 g animal^{-1} d^{-1} (Table 1). Using similar instrumentation, Leytem et al. (2011) found that annualized emissions from a 10,000 cow open-lot dairy in Idaho were 10 g animal^{-1} d^{-1} in a study conducted from March to November 2008. The highest emissions (~36 g N_2O animal^{-1} d^{-1}) were observed in spring (April to late May), when the manure in pens was drying and air temperatures were increasing (average temperature, 21°C).

Other studies (Rahman et al., 2013; Zhu et al., 2014) used different methods (GC coupled with inversion modeling and a custom wind tunnel, respectively) and found emission rates ranging from 17 to 38 g animal^{-1} d^{-1} from a North Dakota feedyard and an open-lot dairy in China. Costa et al. (2014) reported much lower N_2O–N emissions from a Brazilian beef cattle feedyard that averaged 0.02 g animal^{-1} d^{-1}; however, spatial variations ranged between 55 and 123%. In this study, the highest N_2O fluxes occurred from recent urine spots. Waldrip et al. (2015c) conducted eight studies with NFT-NSS chambers in two Texas feedyards and found low, but highly variable, N_2O flux rates that ranged from below detectable levels to 3.1 g animal^{-1} d^{-1}.

We calculated daily per capita N_2O emissions from published data obtained from constant EF, model simulations, chambers, and MMs, which averaged 18 ± 10 g N_2O animal^{-1} d^{-1} (range, 0.04–67 g N_2O animal^{-1} d^{-1}) (Fig. 3). This variation was due to inconsistency in the measurement technique used and spatial and temporal irregularity in N_2O production and release attributable to management, animal diet, and environmental conditions. Calculated N_2O fluxes in studies that used chamber-type methods were very low (0.15 ± 0.28 g N_2O animal^{-1} d^{-1}) (Fig. 3a and 3b) compared with those that used MMs (25.4 ± 25.6 g N_2O animal^{-1} d^{-1}) (Fig. 3a). Further study and comparison of differing methodologies on measured and calculated N_2O flux rates from feedyards are needed.

Diel differences in N_2O emissions have been reported from open-lot dairies during summer and fall in eastern China (Zhu et al., 2014), where fluxes tended to be highest at mid-day and lowest in early morning (Fig. 6). This pattern was likely due to increased manure temperature and consequent enhanced N_2O solubility (Blackmer et al., 1982). However, increased N_2O emission at higher temperatures was not observed by Leytem et al. (2011) on an Idaho open-lot dairy, from feces and urine in controlled temperature incubators (Pereira et al., 2012), or on a Kansas feedyard (Aguilar et al., 2014). Waldrip et al. (2015c) reported that feedyard N_2O collected from NFT-NSS chambers was positively related to manure temperature. However, relationships between N_2O fluxes and numerous other variables were identified, including manure H_2O, OM, NO_3^-, NH_4^+, and dissolved N and C concentrations as well as spectral characteristics related to OM humification. Thorough manure characterization could improve understanding of the driving forces for feedyard N_2O production and emission.

Rahman et al. (2013) used a portable wind tunnel at a North Dakota feedyard, where atmospheric N_2O concentrations were highest in spring (March; 0.99 ppm N_2O) and lowest in early

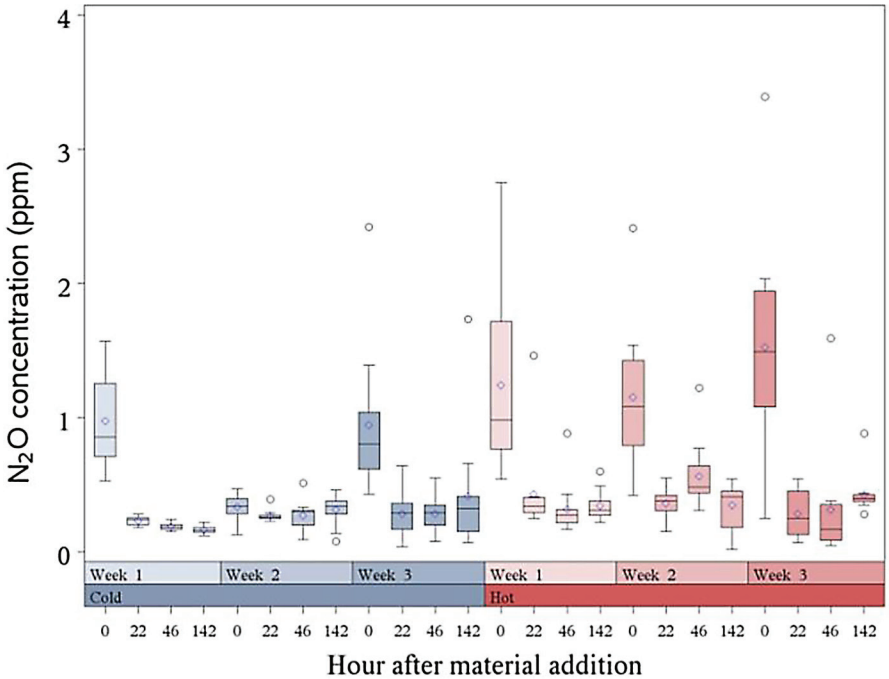

Fig. 5. Concentrations of N_2O in headspace of nonflow-through/nonsteady-state chambers inserted into simulated beef manure packs. Feces, urine, and bedding were incubated for three consecutive 142-h periods under cold (10°C) or hot (40°C) temperatures. Manure moisture content was maintained at 70% throughout the experiment. This work showed that most N_2O was produced immediately after addition of fresh material and that temperature played a major role in the magnitude of N_2O production. Source: Ayadi et al. (2015).

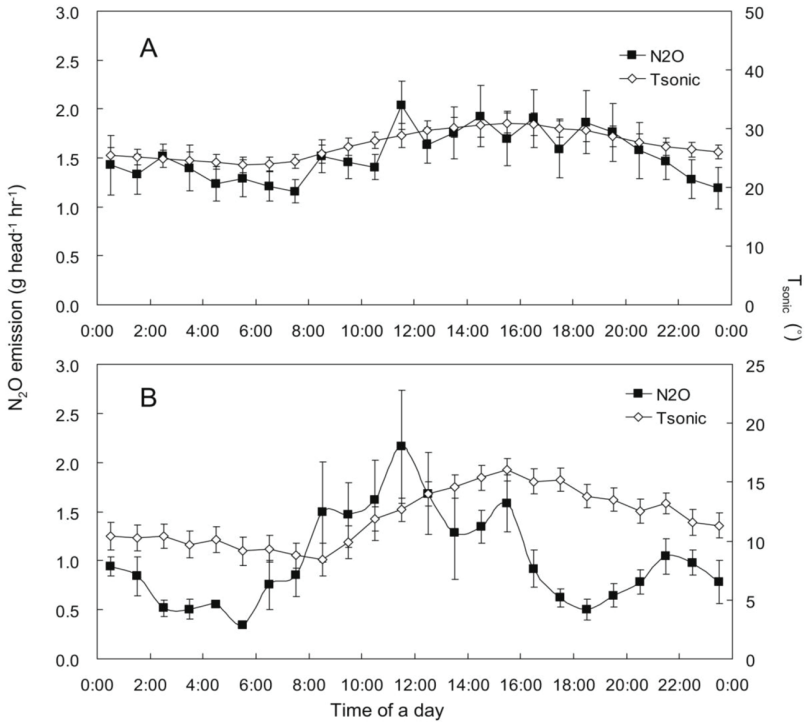

Fig. 6. Diurnal patterns of N_2O emissions from (A) milking cows and (B) heifers on an open-lot dairy in China. Temperature (Tsonic; °C) was more closely related to N_2O fluxes for milking cows than heifers. Source: Zhu et al. (2014).

summer (June) and winter (January; ~0.45 ppm N_2O). Manure H_2O content was proposed as the likely cause of this observation because drier conditions facilitated nitrification in spring. However, it should be noted that estimated fluxes were ~46 times higher than those measured from dynamic chambers on a Texas feedyard by Borhan et al. (2011). Rahman et al. (2013) attributed this contrast between studies to the specific measurement method used and questioned the suitability of flux chambers for feedyard N_2O measurements.

Most inventory calculations and models for estimating N_2O emission include information regarding the amount of manure N excreted; however, Redding et al. (2015) and Waldrip et al. (2015c) found no relationship between N_2O emissions and NO_3^- or total N concentrations in feedyard manure. Redding et al. (2015) measured

N_2O emissions from Australian feedyards using a large chamber and a Fourier-transform infrared monitor. Fluxes of N_2O increased with manure density, pH, temperature, and manure mass and decreased with manure H_2O and organic C content. In short, the interplay among factors involved in N_2O production and emission processes considerably complicates quantitative analyses of the role of feedyards in global climate change. Continued research is needed at the feedyard scale and in isolated micro-sites (i.e., small chamber and laboratory incubation studies) to supplement current understanding of feedyard N_2O.

Estimating Feedyard N_2O Emissions

Estimates of N_2O emission from CAFOs are required for regional and national GHG assessments and may soon be required for regulatory reporting purposes. Improved emission estimates will assist in evaluation of mitigation practices for feedyards and will help to forecast changes in N_2O emission due to changing climate and management. The USEPA and IPCC use constant EF to estimate N_2O production (0.02 kg N_2O–N kg^{-1} manure N excreted for open-lot systems) (IPCC, 2006). Deep-bedded cattle housing (e.g., monoslope barns) were expected to produce half the N_2O emitted from open-lot pens, with an EF of 0.01 kg N_2O–N kg^{-1} N excreted. However, there were large variations in the data used to develop these EFs; thus, uncertainty is high (−16% to +24%) (USEPA, 2013). One key limitation of EFs is a failure to account for changes in microbial and chemical processes due to temperature, H_2O, and other factors affecting N_2O production and emission.

Process-based models provide an alternative that better represent site-specific conditions and interactions among factors influencing emissions. Several models are available to estimate soil N_2O emissions. They include, but are not limited to, the Daily Century Model (DayCent) (Del Grosso et al., 2006), the Denitrification and Decomposition Model (DNDC) (Li, 2007; Li et al., 1992), and the Integrated Farm Systems Model (IFSM) (Chianese et al., 2009; Rotz et al., 2012). These models simulate soil processes to predict N_2O release based on environmental conditions (e.g., temperature, precipitation), management (e.g., fertilization regime, land use type, crop growth), and soil properties (e.g., pH, nutrient concentrations, bulk density, porosity).

Despite progress in modeling soil N_2O emissions, considerably less work has been conducted to develop and improve process-based models for predicting N_2O emissions from open-lot CAFOs. Li et al. (2012) added a manure component to DNDC, calling the new model Manure-DNDC. In Manure-DNDC, denitrification is based on biochemical reaction kinetics as a function of denitrifier growth rate, mass balance between N_2O production and consumption, and gas diffusion through manure. Nitrification rate is predicted according to Michaelis–Menten kinetics, where nitrifier activity is a function of dissolved organic C and NH_4^+ concentrations, temperature, redox potential, and pH in manure. Enteric N_2O is modeled as a linear function of daily N consumed in feed, diet composition, and animal growth and milk production.

Costa et al. (2014) evaluated Manure-DNDC predictions against measured data from a Brazilian feedyard. Manure-DNDC predictions were within the standard deviations of measured N_2O emissions, and model predictions mirrored measured changes in N_2O fluxes due to precipitation and temperature. However, Manure-DNDC tended to underestimate when measured N_2O emissions were high, perhaps due to overestimation of NO_3^- leaching through manure. Li et al. (2012) evaluated Manure-DNDC predictions against 2 yr of measured data where cattle slurries were applied to soil in Scotland. Manure-DNDC effectively simulated peak N_2O fluxes after slurry application and differences in N_2O emissions due to C to N ratios of the applied slurries.

The IFSM is another process-based model that has been evaluated for open-lot systems (Rotz et al., 2012). Nitrification and denitrification are predicted with IFSM using relationships from the DayCent soil model (Del Grosso et al., 2006). Factors considered in estimates of N_2O emissions include manure NH_4^+, NO_3^-, and H_2O concentrations; porosity and pH; and air temperature. The IFSM has been used successfully to evaluate potential environmental impacts of different dairy management systems, where N_2O emission estimates varied with climate and agronomic practices (Belflower et al., 2012; Bonfacio et al., 2015; Stackhouse-Lawson et al., 2013).

Bonfacio et al. (2015) predicted mean daily per capita N_2O emission of 14 g $animal^{-1}$ at an Idaho open-lot dairy, as compared with mean measured emission of 17 g N_2O $animal^{-1}$ by Leytem et al. (2011) (Table 1; Fig. 7). Overall, Bonfacio et al. (2015) saw 80% agreement between IFSM predictions and measured emissions. The IFSM accurately simulated changes in denitrification rates due to manure H_2O content. However, IFSM evaluation and improvement has been limited by low availability of long-term N_2O emission data from open-lot systems.

Collaboration between model developers and researchers measuring emissions, both at the feedyard scale and under controlled laboratory conditions, will assist in improving the parameterization and evaluation of models for estimation of feedyard N_2O emissions. Valid process-based models will be useful tools to quantify N_2O fluxes from open-lot cattle systems, allowing feedyard operators to make management decisions that minimize N_2O volatilization from manure.

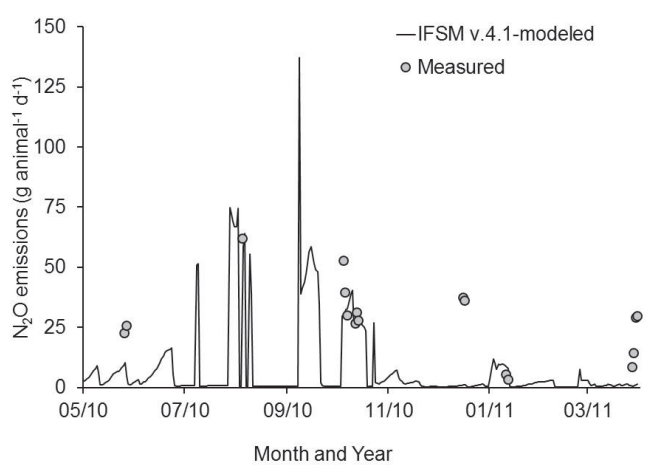

Fig. 7. Measured and Integrated Farm System Model (IFSM)-predicted N_2O emissions for an Idaho dairy from May 2010 to March 2011. Average measured emission rates were 17 g N_2O $animal^{-1}$ d^{-1} and IFSM predicted rates were 14 g N_2O $animal^{-1}$ d^{-1}, with RMSE of 13 g N_2O $animal^{-1}$ d^{-1} and index of agreement of 80%. Source: Bonfacio et al. (2015).

Mitigation of Feedyard N_2O Emissions

Increasing Cattle Productivity and Reducing Dietary Protein Intake

Feedyard N_2O emission is related to animal productivity, where higher-producing cattle consume more feed and excrete greater quantities of manure N than lower-producing animals. However, as meat and milk production increases, N_2O losses per unit of marketable product are reduced. There is less cumulative N_2O production from high-producing cattle than from lower-producing cattle when expressed as emission intensity (N_2O emitted per unit of animal product). Currently, increasing animal productivity is the most viable means of reducing feedyard N_2O emission. Cattle productivity can be increased through improvements in genetics, feeding, health, and management. As an example, Hristov et al. (2013) estimated that total GHG emissions were 70% lower from cattle gaining 1.5 versus 1.0 kg body weight d^{-1}.

Ionophores, anabolic implants (estrogens, testosterones, and progestins), antibiotics, and β-agonists (nonhormonal compounds that reduce fat metabolism) are often used at conventionally managed US feedyards to improve animal productivity and to prevent disease (Wileman et al., 2009). These treatments reduce beef cattle N_2O emission intensity by increasing growth rate and feed efficiency, resulting in a shorter time-frame to reach target finishing weight. Anabolic implants increase final body weight while maintaining a similar body composition to unimplanted animals (Guiroy et al., 2002). Ionophores and antibiotics improve feed efficiency and animal health, whereas β-agonists fed during the final phases of finishing improve weight gain efficiency and increase lean tissue gain.

Other than improved animal efficiency, it remains unclear how growth-promoting treatments influence feedyard N_2O emission. Stackhouse-Lawson et al. (2013) evaluated performance, carcass traits, and N_2O emissions from steers receiving different growth-promoting agents. Overall, there was a 7 to 20% increase in N_2O emissions from cattle that received zilpatrol hydrochloride, monensin, and tylosin phosphate compared with unsupplemented cattle; however, N_2O emission intensity decreased when cattle received multiple growth-promoting compounds. Hristov et al. (2013) suggested that growth-promoting compounds may have little effect on feedyard emissions. However, improved animal health results in less cattle mortality and should decrease the feedyard N_2O contribution per unit of saleable product.

Montes et al. (2013) reviewed methods for mitigating manure-derived N_2O emissions and recommended dietary manipulation to improve the balance between nutrient inputs, production, and environmental impact. Diet composition directly affects N_2O emission by influencing N excretion characteristics and manure microbial composition. The manure from dairy cattle fed diets containing a high proportion of forage tends to contain more fungi and emit less N_2O than manure from cattle consuming a high protein/low fiber diet (Jost et al., 2013). Hünerberg et al. (2014) conducted a life-cycle analysis and estimated that inclusion of WDGS at 40% of daily DM intake slightly increased N_2O emissions from 21 to 22% of total CO_2eq in finishing cattle. Distillers' grains contain approximately 30 to 40% CP, as compared with approximately 10% CP for steam-flaked corn (NRC, 2000). The authors attributed higher predicted N_2O emissions to excessive N consumption in relation to cattle requirements. The importance of cattle diet on N excretion was recently reviewed by Waldrip et al. (2015a). Hales et al. (2013) reported that inclusion of 45% WDGS in cattle diets increased total N excretion by 47%. Both total N intake and diet composition are important considerations when assessing the N sustainability of cattle operations.

Feedyard Pen Management to Mitigate N_2O Emissions

Managing Manure H_2O Content

Nitrous oxide emissions can be reduced when gravimetric H_2O content of the feedyard pen is maintained within 20 to 40% of manure DM; however, alternative pen designs that evenly distribute manure and H_2O might be required for this practice to be effective (Miller and Berry, 2005). Aguilar et al. (2014) investigated Kansas feedyard N_2O fluxes from areas designated as moist/muddy, dry and loose, dry and compacted, and wet/flooded. Moist/muddy areas were found to be "hot spots" for microbial activity, with median emission rates of 2.03 mg N_2O m^{-2} h^{-1}, whereas N_2O fluxes from dry or flooded areas were negligible (Table 1). Flooded conditions can reduce N_2O emissions due to lowered redox potential (Johnson-Beebout et al., 2009) and restricted gas diffusion through the manure pack.

Chemical Amendments

Urease inhibitors, nitrification inhibitors, and other chemical amendments have been evaluated for reducing N_2O emissions from fertilized fields, but research on their use with CAFOs and grazing livestock has been relatively limited, and results have been highly variable (de Klein and Monaghan, 2011; Gerber et al., 2013; Subbarao et al., 2006). *N*-(*n*-butyl) thiophosphoric triamide (NBPT) is a urease inhibitor used on fertilized cropland. Urease inhibitors delay urea hydrolysis to NH_4^+. This extends the time available for root uptake of urea and reduces the opportunity for nitrification of NH_4^+ to NO_3^-, where N_2O may be produced as an intermediate. The probable mechanism of NBPT inhibition is chelation with nickel in the active site of the urease enzyme (Blakeley and Zerner, 1984; Upadhyay, 2012). In cattle pens and manure, NBPT application can reduce NH_3 emissions (Parker et al., 2005, 2012; Shi et al., 2001). However, little is known regarding the effects of NBPT on feedyard N_2O emissions. It is possible that NBPT use in feedyards could lead to "pollution swapping," where NBPT-treated manure with higher total N content would emit more N_2O and less NH_3 than manure that did not receive NBPT application. Further research is warranted in this area.

Nitrification inhibitors directly block the nitrification pathway, limiting substrates for denitrification and reducing N_2O losses from urine spots deposited by cattle (Morales et al., 2015). The most commonly used nitrification inhibitors for agricultural soils are 2-chloro-6-(trichloromethyl) pyridine (nitrapyrin), dicyandiamide (DCD), and 3,4-dimethylpyrazole phosphate (DMPP). It is proposed that DCD specifically inhibits NH_3 oxidation by blocking the conversion of NH_3 to NH_2OH (Di et al., 2009), which is the first step in nitrification (Fig. 2). Morales et al. (2015) observed that DCD applied at a rate of 10 mg kg^{-1} (DM) to urine-amended soils reduced N_2O emissions by 23 to 67%. No definitive changes with DCD application were observed in the soil community composition of nitrifiers, potential denitrifiers, or organisms involved in alternative N-utilizing pathways. Other researchers have studied the efficacy of DCD and DMPP on mitigating N_2O emission from urine spots in grazed pasture,

with mixed results (Luo et al., 2013; Smith et al., 2008; Zaman and Nguyen, 2012). Mazzetto et al. (2015) investigated DCD for mitigation of N_2O from cattle urine on an acidic Brazilian Oxisol (pH 5.0) and found no benefit from DCD application.

Very little research has evaluated DCD or other nitrification inhibitors on N_2O from open-lot feedyards. The feedyard manure microbial community composition, constant excretion of urine and feces onto the pen floor, and high manure OM content may challenge nitrification inhibitors used successfully on fertilized cropland. Sahrawat et al. (1987) reported that both DCD and nitrapyrin slowed nitrification rates in a silt loam but were ineffective in an organic Histosol. Calderon et al. (2005) reported that nitrapyrin reduced nitrification but had inconsistent effects on N_2O emission from manured soils. Nitrapyrin delayed nitrification in anhydrous NH_3–fertilized corn plots and temporarily reduced N_2O emissions in a study by Parkin and Hatfield (2010); however, cumulative annual N_2O losses were not significantly different from plots that did not receive nitrapyrin. Research is needed to evaluate long-term cost effectiveness, optimal application rate, and required application frequency of chemical inhibitors as well as their potential toxicity to cattle and feedyard personnel and overall practicality for mitigation of feedyard N_2O emissions.

Organic and Other Amendments

The addition of biochar or other organic C sources to feedyard pens may reduce N_2O emissions (Aguilar et al., 2014). One proposed mechanism of N_2O inhibition after organic residue application is via sorption and retention of N_2O, NH_4^+, and/or NO_3^- (Aguilar et al., 2014; Yao et al., 2012). However, the effectiveness of organic amendments is influenced by manure H_2O content, amendment application rate, and environmental temperature (Aguilar et al., 2014). Aguilar et al. (2014) observed 57 to 73% reduction in N_2O emission when manure biochars or activated C was added to moist (35% H_2O) beef cattle manure in a short-term laboratory study. However, the high amendment rate required to reduce GHG losses in this study (1.1–5.1 kg m^{-2}) may not be practical or cost-effective for commercial feedyards. In this study, temperatures above 40°C caused N_2O desorption from amendments, leading to larger N_2O fluxes at high temperatures. In short, organic amendments may be ineffective for mitigation of N_2O emissions during the dry, hot summers typical of Texas and other major beef-finishing regions in the United States.

Amendments used to control emissions of NH_3 and other compounds in CAFOs, including zeolite, lignite, bedding or other organic material, aluminum sulfate, tannins, acidifying compounds, and biochar, have yet to be evaluated for open-lot feedyards. This is an area that requires significant research regarding environmental impact, practicality, cost-effectiveness, and animal and human health effects before making producer-oriented suggestions for their use on commercial feedyards.

Conclusions and Areas for Further Research

A thorough literature review revealed that published per capita flux rates for open-lot cattle systems were highly variable and ranged from 0.002 to 4.3 kg N_2O $animal^{-1}$ yr^{-1} or 0.02 to 37 g N_2O $animal^{-1}$ d^{-1}. On an area basis, published emissions rates from feedyards ranged from 0 to 41 mg N_2O m^{-2} h^{-1}. Much of this variation was due to inconsistency in the measurement technique used and to distinct spatial and temporal irregularity in N_2O production and volatilization due to management practices, animal diet, and environmental conditions. Feedyard N_2O emissions appear to be largely controlled by environmental temperature and manure N and H_2O content; however, relationships were also identified between animal diet, pen management, and animal stocking density. During the preparation of this review, five primary knowledge gaps were identified where current understanding is weak and further research is required: (i) accurate measurement of N_2O emissions from feedyards using appropriate, and when available, standardized methodologies; (ii) improved understanding of the microbiology, chemistry, and physical structure of pen surface layers that lead to N_2O emissions; (iii) improved understanding of factors that influence feedyard N_2O emissions, including manure H_2O content, porosity, density, available N and C contents, environmental temperatures, and use of veterinary pharmaceuticals; (iv) development of cost-effective and practical mitigation strategies to decrease N_2O emissions from pen surfaces, manure stockpiles, composting windrows, and retention ponds; and (v) improved process-based models that can accurately predict feedyard N_2O emissions, evaluate mitigation strategies, and forecast future N_2O emission trends.

Given the impetus toward increasing regulation, feedyard managers, nutritionists, and researchers may play increasingly important roles in feedyard N management. Common feedyard management practices may need modification or refinement to balance production efficiency with environmental concerns. Large-scale micrometerological measurement campaigns and small-scale chamber studies are needed to assess the overall magnitude of feedyard N_2O emissions and to determine key factors driving N_2O production and emission.

References

Aguilar, O., R. Maghirang, C.W. Rice, S. Trabue, L.E. Erickson, and E. Razote. 2011. Nitrous oxide emissions from a commercial cattle feedlot in Kansas. Proceedings of the 2011 ASABE Annual International Meeting, Louisville, KY. 7–10 Aug. 2011. ASABE, St. Joseph, MI.

Aguilar, O.A., R. Maghirang, C.W. Rice, and S.L. Trabue. 2014. Nitrous oxide fluxes from a commercial beef cattle feedlot in Kansas. Air Soil Water Res. 7:35–45. doi:10.4137/ASWR.S12841

Amarakoon, I.D., F. Zvomuya, S. Sura, F.J. Larney, A.J. Cessna, S. Xu, et al. 2016. Dissipation of antimicrobials in feedlot manure compost after oral administration versus fortification after excretion. J. Environ. Qual. 45:503–510. doi:10.2134/jeq2015.07.0408

Antileo, C., E. Aspe, H. Urrutia, C. Zaror, and M. Roeckel. 2002. Nitrifying biomass acclimation to high ammonia concentration. J. Environ. Eng. 128:367–375. doi:10.1061/(ASCE)0733-9372(2002)128:4(367)

Ayadi, F.Y., E.L. Cortus, M.J. Spiehs, D.N. Miller, and G.D. Djira. 2015. Ammonia and greenhouse gas concentrations at surfaces of simulated beef cattle bedded manure packs. Trans. ASABE 58:783–795. doi:10.13031/trans.58.10961

Baum, K.A. 2008. Air emissions at cattle feedlots. M.S. thesis. Kansas State Univ., Manhattan. http://krex.k-state.edu/dspace/bitstream/handle/2097/775/BaumKristen2008.pdf?sequence=1&isAllowed=y (accessed 13 Jan. 2016).

Baum, K.A., and J.M. Ham. 2009. Adaptation of a speciation sampling cartridge for measuring ammonia flux from cattle feedlots using relaxed eddy accumulation. Atmos. Environ. 43:1753–1759. doi:10.1016/j.atmosenv.2008.12.021

Belflower, J.B., J.K. Bernard, D.K. Gattie, D.W. Hancock, L.M. Rise, and C.A. Rotz. 2012. A case study of the potential environmental impacts of different dairy production systems in Georgia. Agric. Syst. 108:84–93. doi:10.1016/j.agsy.2012.01.005

Blackmer, A.M., S.G. Robbins, and J.M. Bremner. 1982. Diurnal variability in the rate of emission of nitrous oxide from soils. Soil Sci. Soc. Am. J. 46:937–942. doi:10.2136/sssaj1982.03615995004600050011x

Blakeley, R.L., and B. Zerner. 1984. Jack bean urease: The first nickel enzyme. J. Mol. Catal. 23:263–292. doi:10.1016/0304-5102(84)80014-0

Blasing, T.J. 2014. Recent greenhouse gas concentrations. Carbon Dioxide Information Analysis Center, Oak Ridge National Laboratory, Oak Ridge, TN. http://cdiac.ornl.gov/pns/current_ghg.html (accessed Sept. 2015).

Boadi, D.A., K.M. Wittenberg, S.L. Scott, D. Burton, K. Buckley, J.A. Small, et al. 2004. Effect of low and high forage diet on enteric and manure pack greenhouse gas emissions from a feedlot. Can. J. Anim. Sci. 84:445–453. doi:10.4141/A03-079

Bonfacio, H.F., C.A. Rotz, A.B. Leytem, H.M. Waldrip, and R.W. Todd. 2015. Process-based modeling of ammonia and nitrous oxide emissions from open-lot beef and dairy facilities. Trans. ASABE 58:827–846. doi:10.13031/trans.58.11112

Borhan, M.S., S.C. Capareda, S. Mukhtar, W.B. Faulkner, R. McGee, and C.B. Parnell. 2011. Greenhouse gas emissions from ground level area sources in dairy and cattle feeding operations. Atmosphere 2:303–329. doi:10.3390/atmos2030303

Brady, N.C., and R.C. Weil. 2002. The nature and properties of soils. 13th ed. Prentice Hall, Upper Saddle River, NJ.

Bremner, J.M., and K. Shaw. 1958. Denitrification in soil: II. Factors affecting denitrification. J. Agric. Sci. 51:40–52. doi:10.1017/S0021859600032779

Brown, H.A., C. Wagner-Riddle, and G.W. Thurtell. 2000. Nitrous oxide flux from solid dairy manure in storage as affected by water content and redox potential. J. Environ. Qual. 29:630–638.

Butterbach-Bahl, K., E.M. Baggs, M. Dannenmann, R. Kiese, and S. Zechmeister-Boltenstern. 2013. Nitrous oxide emissions from soils: How well do we understand the processes and their controls? Philos. Trans. R. Soc. B. 368:20130122. doi:10.1098/rstb.2013.0122

Calderon, F.J., G.W. McCarty, and J.B. Reeves, III. 2005. Nitrapyrin delays denitrification on manured soils. Soil Sci. 170:350–359. doi:10.1097/01.ss.0000169905.94861.c7

Cassel, T., L. Ashbaugh, R. Flocchini, and D. Meyer. 2005. Ammonia emission factors for open-lot dairies: Direct measurements and estimation by nitrogen intake. J. Air Waste Manage. Assoc. 55:826–833. doi:10.1080/10473289.2005.10464660

Chianese, D.S., C.A. Rotz, and T.L. Richard. 2009. Whole-farm greenhouse gas emissions: A review with application to a Pennsylvania dairy farm. Appl. Eng. Agric. 25:431–442. doi:10.13031/2013.26895

Cole, N.A., and R.W. Todd. 2009. Nitrogen and phosphorus balance of beef cattle feedyards. In: Proceedings of the Texas Animal Manure Management Issues Conference, Round Rock, TX. 29–30 Sept. 2009. Round Rock, TX. USDA–ARS, Washington, DC. p. 17–24.

Cole, N.A., R.W. Todd, D.B. Parker, and M.B. Rhoades. 2007. Challenges in using flux chambers to measure ammonia emissions from simulated feedlot pen surfaces and retention ponds. Proceedings of the International Symposium on Air Quality and Waste Management for Agriculture, Broomfield, CO. 16–19 Sept. 2007. ASABE, St. Joseph, MI.

Cole, N.A., A.M. Mason, R.W. Todd, and D.B. Parker. 2009a. Effects of urine application on chemistry of feedlot pen surfaces. J. Anim. Sci. 87. E Suppl. 2:148.

Cole, N.A., A.M. Mason, R.W. Todd, M. Rhoades, and D.B. Parker. 2009b. Chemical composition of pen surface layers of beef cattle feedyards. Prof. Anim. Sci. 25:541–552.

Cortus, E.L., L.D. Jacobson, B.P. Hetchler, A.J. Heber, and B.W. Bogan. 2015. Methane and nitrous oxide analyzer comparison and emissions from dairy freestall barns with manure flushing and scraping. Atmos. Environ. 100:57–65. doi:10.1016/j.atmosenv.2014.10.039

Costa, C., C. Li, C.P. Cerri, and C.C. Cerri. 2014. Measuring and modeling nitrous oxide and methane emissions from beef cattle feedlot manure management: First assessments under the Brazilian condition. J. Environ. Sci. Health B 49:696–711. doi:10.1080/03601234.2014.922856

Dalal, R.C., W. Wang, G.P. Robertson, and W.J. Parton. 2003. Nitrous oxide emission from Australian agricultural lands and mitigation options: A review. Aust. J. Soil Res. 41:165–195. doi:10.1071/SR02064

de Klein, C.A., and R.M. Monaghan. 2011. The effect of farm and catchment management on nitrogen transformations and N_2O losses from pastoral systems: Can we offset the effect of future intensification? Curr. Opin. Environ. Sustainability 3:396–406. doi:10.1016/j.cosust.2011.08.002

Del Grosso, S.J., W.J. Parton, A.R. Mosier, M.K. Walsh, D.S. Ojima, and P.E. Thornton. 2006. DayCent national scale simulations of N_2O emissions from cropped soils in the USA. J. Environ. Qual. 35:1451–1460. doi:10.2134/jeq2005.0160

Di, H.J., K.C. Cameron, R.R. Sherlock, C.S. Winefield, M. O'Callaghan, S. Bowatte, and J.Z. He. 2009. Nitrification driven by bacteria and not archaea in nitrogen-rich grassland soils. Nat. Geosci. 2:621–624. doi:10.1038/ngeo613

Di, H.J., K.C. Cameron, R.R. Sherlock, J.P. Shen, J.-Z. He, and C.S. Winefield. 2010. Nitrous oxide emissions from grazed grassland as affected by nitrification inhibitor, dicyandiamide, and relationships with ammonia-oxidizing bacteria and archaea. J. Soils Sediments 10:943–954. doi:10.1007/s11368-009-0174-x

Elanco. 2005. Finding of no significant impact for Rumensin (monensin sodium) 80 Type A medicated article for cattle fed in confinement at 40 g/ton (480 mg/head/day). Elanco Animal Health, Greenfield, IN. http://www.fda.gov/downloads/AnimalVeterinary/DevelopmentApprovalProcess/EnvironmentalAssessments/UCM324418.pdf (accessed 8 Sept. 2016).

Elmund, G.K., S.M. Morrison, D.W. Grant, and M.P. Nevins. 1971. Role of excreted chlortetracycline in modifying the decomposition process in feedlot waste. Bull. Environ. Contam. Toxicol. 6:129–132. doi:10.1007/BF01540093

Firestone, M.K., and E.A. Davidson. 1989. Microbiological basis of NO and N2O production and consumption in soil. In: M.O. Andreae and D.S. Schimel, editors, Exchange of trace gases between terrestrial ecosystems and the atmosphere. John Wiley & Sons, New York. p. 7–21.

Flesch, T.K., L.A. Harper, J.M. Powell, and J.D. Wilson. 2009. Inverse-dispersion calculation of ammonia emissions from Wisconsin dairy farms. Trans. ASABE 52:253–265. doi:10.13031/2013.25946

Fortuna, A.M., C.W. Honeycutt, G. Vandemark, T.S. Griffin, R.P. Larkin, Z. He, B.J. Weinhold, et al. 2012. Links among nitrification, nitrifier communities, and edaphic properties in contrasting soils receiving dairy slurry. J. Environ. Qual. 41:262–272. doi:10.2134/jeq2011.0202

Galbally, I.E. 1989. Factors controlling NO_x emissions from soils. In: M.O. Andreae and D.S. Schimel, editors, Exchange of trace gases between terrestrial ecosystems and the atmosphere. Biddles Ltd., Guildford, UK. p. 23–37.

Gerber, P.J., A.N. Hristov, B. Henderson, H. Makkar, J. Oh, C. Lee, et al. 2013. Technical options for the mitigation of direct methane and nitrous oxide emissions from livestock: A review. Animal 7:220–234. doi:10.1017/S1751731113000876

Gruber, N., and J.N. Galloway. 2008. An Earth-system perspective of the global nitrogen cycle. Nature 451:293–296. doi:10.1038/nature06592

Guiroy, P.J., L.O. Tedeschi, D.G. Fox, and J.P. Hutcheson. 2002. The effects of implant strategy on finished body weight of beef cattle. J. Anim. Sci. 80:1791–1800.

Hales, K.E., N.A. Cole, and J.C. MacDonald. 2013. Effects of increasing concentrations of wet distillers grains with solubles in steam-flaked, corn-based diets on energy metabolism, carbon-nitrogen balance, and methane emissions of cattle. J. Anim. Sci. 91:819–828. doi:10.2527/jas.2012-5418

Hao, X., S. Xu, J. Larney, K. Stanford, A.J. Cessna, and T.A. McAllister. 2011. Inclusion of antibiotics in feed alters greenhouse gas emissions from feedlot manure during composting. Nutr. Cycling Agroecosyst. 89:257–267. doi:10.1007/s10705-010-9391-3

Hatzenpichler, R. 2012. Diversity, physiology, and niche differentiation of ammonia-oxidizing archaea. Appl. Environ. Microbiol. 78:7501–7510. doi:10.1128/AEM.01960-12

Hensen, A., U. Skiba, and D. Famulari. 2013. Low cost and state of the art methods to measure nitrous oxide emissions. Environ. Res. Lett. 8:025022. doi:10.1088/1748-9326/8/2/025022

Herrmann, M., A.M. Saunders, and A. Schramm. 2008. Archaea dominate the ammonia-oxidizing community in the rhizosphere of the freshwater macrophyte *Littorella uniflora*. Appl. Environ. Microbiol. 74:3279–3283. doi:10.1128/AEM.02802-07

Hristov, A.N., J. Oh, J.L. Firkins, J. Dijkstra, E. Kebreab, G. Waghorn, et al. 2013. Mitigation of methane and nitrous oxide emissions from animal operations: I. A review of enteric methane mitigation options. J. Anim. Sci. 91:5045–5069. doi:10.2527/jas.2013-6583

Hünerberg, M., S.M. Litte, K.A. Beauchemin, S.M. McGinn, D. O'Connor, E.K. Okine, et al. 2014. Feeding high concentrations of corn dried distillers' grains decreases methane, but increases nitrous oxide emissions from beef cattle production. Agric. Syst. 127:19–27. doi:10.1016/j.agsy.2014.01.005

Hutchinson, G.L., and G.P. Livingston. 2001. Vents and seals in nonsteady-state chambers used for measuring gas exchange between soil and the atmosphere. Eur. J. Soil Sci. 52:675–682. doi:10.1046/j.1365-2389.2001.00415.x

Hutchinson, G.L., and G.P. Livingston. 2002. Soil-atmosphere gas exchange. In: J.H. Dane and G.C. Topp, editors, Methods of soil analysis. Part 4. SSSA Book Ser. 5. SSSA, Madison, WI. p. 1159–1182.

Hutchinson, G.L., and A.R. Mosier. 1981. Improved soil cover method for field measurement of nitrous oxide fluxes. Soil Sci. Soc. Am. J. 45:311–316. doi:10.2136/sssaj1981.03615995004500020017x

IPCC. 2006. Emissions from livestock and manure management. In: 2006 Guidelines for National Greenhouse Gas Inventories. Prepared by the National Greenhouse Gas Inventories Programme. IGES, Japan. http://www.ipcc-nggip.iges.or.jp/public/2006gl/pdf/4_Volume4/V4_10_Ch10_Livestock.pdf (accessed 8 Sept. 2016).

Johnson-Beebout, S.E., O.R. Angeles, M.C.R. Alberto, and R.J. Buresh. 2009. Simultaneous minimization of nitrous oxide and methane emission from rice paddy soils is improbable due to redox potential changes with depth in a greenhouse experiment without plants. Geoderma 149:45–53. doi:10.1016/j.geoderma.2008.11.012

Jost, D.I., R.G. Joergensen, and A. Sundrum. 2013. Effect of cattle feces with different microbial biomass content on soil properties, gaseous emissions and plant growth. Biol. Fertil. Soils 49:61–70. doi:10.1007/s00374-012-0697-y

Jungbluth, T., E. Hartung, and G. Brose. 2001. Greenhouse gas emissions from animal houses and manure stores. Nutr. Cycling Agroecosyst. 60:133–145. doi:10.1023/A:1012621627268

Kaspar, H.F., and J.M. Tiedje. 1981. Dissimilatory reduction of nitrate in the bovine rumen: Nitrous oxide production and effect of acetylene. Appl. Environ. Microbiol. 41:705–709 http://aem.asm.org/content/41/3/705.full.pdf+html (accessed 8 Sept. 2016).

Koper, T.E., J.M. Stark, M.Y. Habteselassie, and J.M. Norton. 2010. Nitrification exhibits Haldane kinetics in an agricultural soil treated with ammonium sulfate or dairy-waste compost. FEMS Microbiol. Ecol. 74:316–322. doi:10.1111/j.1574-6941.2010.00960.x

Laanbroek, H.J., and S. Gerards. 1991. Effects of organic manure on nitrification in arable soils. Biol. Fertil. Soils 12:147–153. doi:10.1007/BF00341492

Laughlin, R.J., T. Rütting, C. Müller, C.J. Watson, and R.J. Stevens. 2009. Effect of acetate on soil respiration, N_2O emissions and gross N transformations related to fungi and bacteria in a grassland soil. Appl. Soil Ecol. 42:25–30. doi:10.1016/j.apsoil.2009.01.004

Lensi, R., and A. Chalamet. 1982. Denitrification in waterlogged soils: In situ temperature-dependent variations. Soil Biol. Biochem. 14:51–55. doi:10.1016/0038-0717(82)90076-1

Leytem, A.B., R.S. Dungan, D.L. Bjorneberg, and A.C. Koehn. 2011. Emissions of ammonia, methane, carbon dioxide, and nitrous oxide from dairy cattle housing and manure management systems. J. Environ. Qual. 40:1383–1394. doi:10.2134/jeq2009.0515

Leytem, A.B., R.S. Dungan, D.L. Bjorneberg, and A.C. Koehn. 2013. Greenhouse gas and ammonia emissions from an open-freestall dairy in southern Idaho. J. Environ. Qual. 42:10–20. doi:10.2134/jeq2012.0106

Levy, P.E., A. Gray, S.R. Leeson, J. Gaiawyn, M.P.C. Kelly, M.D.A. Cooper, et al. 2011. Quantification of uncertainty in trace gas fluxes measured by the static chamber method. Eur. J. Soil Sci. 62:811–821. doi:10.1111/j.1365-2389.2011.01403.x

Li, C. 2007. Quantifying greenhouse gas emissions from soils: Scientific basis and modeling approach. Soil Sci. Plant Nutr. 53:344–352. doi:10.1111/j.1747-0765.2007.00133.x

Li, C., S. Frolking, and T.A. Frolking. 1992. A model of nitrous oxide evolution from soil driven by rainfall events: I. Model structure and sensitivity. J. Geophys. Res. 97:9759–9776. doi:10.1029/92JD00509

Li, C., W. Salas, R. Zhang, C. Krauter, A. Rotz, and F. Mitloehner. 2012. Manure-DNDC: A biogeochemical process model for quantifying greenhouse gas and ammonia emissions from livestock manure systems. Nutr. Cycling Agroecosyst. 93:163–200. doi:10.1007/s10705-012-9507-z

Limmer, A.W., and K.W. Steele. 1982. Denitrification potentials: Measurement of seasonal variation using a short-term anaerobic incubation technique. Soil Biol. Biochem. 14:179–184. doi:10.1016/0038-0717(82)90020-7

Livingston, G.P., G.L. Hutchinson, and K. Spartalin. 2006. Trace gas emission in chambers: A non-steady-state diffusion model. Soil Sci. Soc. Am. J. 70:1459–1469. doi:10.2136/sssaj2005.0322

Luo, J., S.F. Ledgard, and S.B. Lindsay. 2013. Nitrous oxide and greenhouse gas emissions from grazed pastures as affected by use of nitrification inhibitor and restricted grazing regime. Sci. Total Environ. 465:107–114. doi:10.1016/j.scitotenv.2012.12.075

Ma, W.K., R.E. Farrell, and S.D. Siciliano. 2008. Soil formate regulates the fungal nitrous oxide emission pathway. Appl. Environ. Microbiol. 74:6690–6696. doi:10.1128/AEM.00797-08

Maeda, K., S. Toyoda, R. Shimojima, T. Osada, D. Hanajima, R. Morioka, et al. 2010. Source of nitrous oxide emissions during the cow manure composting process as revealed by isotopomer analysis of and *amoA* abundance in betaproteobacterial ammonia-oxidizing bacteria. Appl. Environ. Microbiol. 76:1555–1562. doi:10.1128/AEM.01394-09

Maeda, K., D. Hanajima, S. Toyoda, N. Yoshida, R. Morioka, and T. Osada. 2011. Microbiology of nitrogen cycle in animal manure compost. Microb. Biotechnol. 4:700–709. doi:10.1111/j.1751-7915.2010.00236.x

Matthews, R.A., D.R. Chadwick, A.L. Retter, M.S.A. Blackwell, and S. Yamulki. 2010. Nitrous oxide emissions from small-scale farmland features of UK livestock farming systems. Agric. Ecosyst. Environ. 136:192–198. doi:10.1016/j.agee.2009.11.011

Mazzetto, A.M., A.S. Barneze, B.J. Feigl, J.W. Van Groenigen, O. Oenema, C.A.M. de Klein, et al. 2015. Use of the nitrification inhibitor dicyandiamide (DCD) does not mitigate N_2O emission from bovine urine patches under Oxisol in Northwest Brazil. Nutr. Cycling Agroecosyst. 101:83–92. doi:10.1007/s10705-014-9663-4

McCullough, M.C., D.B. Parker, C. Robinson, and B.W. Auvermann. 2001. Hydraulic conductivity, bulk density, moisture content and electrical conductivity of a new sandy loam feedlot surface. Appl. Eng. Agric. 17:539–544. doi:10.13031/2013.6471

Mielke, L.N., and A.P. Mazurak. 1976. Infiltration of water on a cattle feedlot. Trans. ASAE 19:341–344. doi:10.13031/2013.36024

Miller, D.N., and E.D. Berry. 2005. Cattle feedlot soil moisture and manure content: I. Impacts on greenhouse gases, odor compounds, nitrogen losses, and dust. J. Environ. Qual. 34:644–655. doi:10.2134/jeq2005.0644

Monaghan, R.M., and D. Barraclough. 1992. Some chemical and physical factors affecting the rate and dynamics of nitrification in urine-affected soil. Plant Soil 143:11–18. doi:10.1007/BF00009124

Montes, F., R. Meinen, C. Dell, A. Rotz, A.N. Hristov, J. Oh, et al. 2013. Special topics- Mitigation of methane and nitrous oxide emissions from animal operations: II. A review of manure management mitigation options. J. Anim. Sci. 91:5070–5094. doi:10.2527/jas.2013-6584

Morales, S.E., N. Jha, and S. Saggar. 2015. Impact of urine and the application of the nitrification inhibitor DCD on microbial communities in dairy-grazed pasture soils. Soil Biol. Biochem. 88:344–353. doi:10.1016/j.soilbio.2015.06.009

Mosier, A., C. Kroeze, C. Nevison, O. Oenema, S. Seitzinger, and O. van Cleemput. 1998. Closing the global N_2O budget: Nitrous oxide emissions through the agricultural nitrogen cycle. Nutr. Cycling Agroecosyst. 52:225–248. doi:10.1023/A:1009740530221

Myhre, G., D. Shindell, F.-M. Bréon, W. Collins, J. Fuglestvedt, J. Huang, et al. 2013. Anthropogenic and natural radiative forcing. In: T.F. Stocker, D. Qin, G.-K. Plattner, M.M.B. Tignor, S.K. Allen, J. Boschung, et al., editors, Climate change 2013: The physical science basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge Univ. Press, Cambridge, UK and New York. http://www.climatechange2013.org/images/report/WG1AR5 _Chapter08 _FINAL.pdf (accessed 10 Mar. 2016).

Nielsen, T.H., L.P. Nielsen, and N.P. Revsbech. 1996. Nitrification and coupled nitrification-denitrification association with a soil-manure interface. Soil Sci. Soc. Am. J. 60:1829–1840.

Norton, J.M. 2008. Nitrification in agricultural soils. In: J.S. Schepers and W.R. Raun, editors, Nitrogen in agricultural systems. Agronomy Monograph No. 49. ASA SSSA CSSA, Madison, WI. p. 173–199. doi:10.2134/agronmonogr49.c6

NCBA. 2015. Directions. Statistics. National Cattleman's Beef Association, Denver, CO. http://www.beefusa.org/CMDocs/BeefUSA/Producer%20Ed/2015%20CattleFax%20section.pdf (accessed 8 Sept. 2016).

National Research Council (NRC). 2000. Nutrient requirements of beef cattle, 7th revised edition. National Academy Press, Washington, DC. http://www.nap.edu/read/9791/chapter/1 (accessed 7 Mar. 2016).

Parker, D.B., S. Pangrangi, L.W. Greene, L.K. Almas, N.A. Cole, M.B. Rhoades, et al. 2005. Rate and frequency of urease inhibitor application for minimizing ammonia emissions from beef cattle feedyards. Trans. ASAE 48:787–793. doi:10.13031/2013.18321

Parker, D.B., N.A. Cole, K.D. Casey, G. Galvin, R. Ormerod, C.S. Paris, et al. 2009. Wind tunnels vs. flux chambers: Area source emission measurements and the necessity for VOC and odor correction factors. Proceedings of the 19th International Clean Air and Environment Conference, Perth, Australia. 6–9 Sept. 2009. Clean Air Society of Australia and New Zealand, Mooroolbark, Australia.

Parker, D.B., M.B. Rhoades, and N.A. Cole. 2012. Effect of urease inhibitor application rate and rainfall on ammonia emissions from beef manure. Trans. ASABE 55:211–218. doi:10.13031/2013.41248

Parker, D., J. Ham, B. Woodbury, L. Cai, M. Spiehs, M. Rhoades, et al. 2013. Standardization of flux chamber and wind tunnel flux measurements for quantifying volatile organic compound and ammonia emissions from area sources at animal feeding operations. Atmos. Environ. 66:72–83. doi:10.1016/j.atmosenv.2012.03.068

Parkin, T.B., and J.L. Hatfield. 2010. Influence of nitrapyrin on N_2O losses from soil receiving fall-applied anhydrous ammonia. Agric. Ecosyst. Environ. 136:81–86. doi:10.1016/j.agee.2009.11.014

Parkin, T.B., and R.T. Venterea. 2010. Chamber-based trace gas flux measurements. In: R.F. Follett, editor, USDA–ARS GRACEnet sampling protocols. USDA–ARS, Washington, DC. http://www.amazonpire.org/PDF/FC2010/Volume%202/Baker%20et%20al_2003_GRACEnet_analyses_protocol.pdf (accessed 8 Sept. 2016).

Parton, W.J., E.A. Holland, S.J. Del Grosso, M.D. Hartman, R.E. Martin, A.R. Mosier, et al. 2001. Generalized model for NOx and N2O emissions from soils. J. Geophys. Res. 106:17403–17419. doi:10.1029/2001JD900101

Paul, J.W., and E.G. Beauchamp. 1989. Effect of carbon constituents in manure on denitrification in soil. Can. J. Soil Sci. 69:49–61. doi:10.4141/cjss89-006

Pedersen, A.R., S.O. Petersen, and K. Schelde. 2010. A comprehensive approach to soil atmosphere trace-gas flux estimation with static chambers. Eur. J. Soil Sci. 61:888–902. doi:10.1111/j.1365-2389.2010.01291.x

Pereira, J., T.H. Misselbrook, D.R. Chadwick, J. Coutinho, and H. Trindade. 2012. Effects of temperature and dairy cattle excreta characteristics on potential ammonia and greenhouse gas emissions from housing: A laboratory study. Biosyst. Eng. 112:138–150. doi:10.1016/j.biosystemseng.2012.03.011

Petersen, S.O., S. Stamatiadis, and C. Christofides. 2004. Short term nitrous oxide emissions from pasture soil as influenced by urea level and soil nitrate. Plant Soil 267:117–127. doi:10.1007/s11104-005-4688-8

Petersen, S.O., A.L.F. Hellwing, M. Brask, O.H. Øjberg, M. Poulsen, Z. Zhu, et al. 2015. Dietary nitrate for methane mitigation leads to nitrous oxide emissions from dairy cows. J. Environ. Qual. 44:1063–1070. doi:10.2134/jeq2015.02.0107

Phetteplace, H., D. Johnson, and A. Seidl. 2001. Greenhouse gas emissions from simulated beef and dairy livestock systems in the United States. Nutr. Cycling Agroecosyst. 60:99–102. doi:10.1023/A:1012657230589

Pihlatie, M., E. Syväsalo, A. Simojoki, M. Esala, and K. Regina. 2004. Contribution of nitrification and denitrification to N_2O production in peat, clay and loamy sand soils under different soil moisture conditions. Nutr. Cycling Agroecosyst. 70:135–141. doi:10.1023/B:FRES.0000048475.81211.3c

Rahman, S., M.S. Borhan, and K. Swanson. 2013. Greenhouse gas emissions from beef cattle pen surfaces in North Dakota. Environ. Technol. 34:1239–1246. doi:10.1080/09593330.2012.743598

Redding, M.R., J. Devereux, F. Phillips, R. Lewis, T. Naylor, T. Kearton, et al. 2015. Field measurement of beef pen manure methane and nitrous oxide reveals a surprise for inventory calculations. J. Environ. Qual. 44:720–728. doi:10.2134/jeq2014.04.0159

Reynolds, C.K., J.A.N. Mills, L.A. Crompton, and D.I. Givens. 2010. Ruminant nutrition regimes to reduce greenhouse gas emissions in dairy cows. In: G.M. Crovetto, editor, Energy and protein metabolism and nutrition. EEAP Publication No. 127. Wageningen Academic Publishers, Wageningen, The Netherlands.

Ribaudo, M., N. Gollehon, M. Aillery, J. Kaplan, R. Johansson, J. Agapoff, et al. 2003. Manure management for water quality: Costs to animal feeding operations of applying manure nutrients to land, Agricultural Economic Report No. 824. USDA Economic Research Service, Washington, DC. http://www.ers.usda.gov/publications/aer824 (accessed 2 Mar. 2016).

Rice, W.C., A.M. Mason, N.A. Cole, and R.N. Clark. 2007. The influence of feedlot pen surface layers on microbial community structure and diversity. Proceedings of the ASABE International Symposium on Air Quality and Waste Management for Agriculture, Broomfield, CO. 16–19 Sept. 2007. ASABE, St. Joseph, MI.

Rotz, C.A., M.S. Corson, D.S. Chianese, F. Montes, S.D. Hafner, and C.U. Coiner. 2012. The integrated farm system model: Reference manual, version 3.6. USDA–ARS Pasture Systems and Watershed Management Research Unit, University Park, PA. https://www.ars.usda.gov/northeast-area/up-pa/pswmru/docs/integrated-farm-system-model/#Reference (accessed 8 Sept. 2016).

Sabey, B.R., W.V. Bartholomew, R. Shaw, and J. Pesek. 1956. Influence of temperature on nitrification in soils. Soil Sci. Soc. Am. Proc. 20:357–360. doi:10.2136/sssaj1956.03615995002000030016x

Saggar, S., R.L. Parfitt, G. Salt, and M.F. Skinner. 1998. Carbon and phosphorus transformations during decomposition of pine forest floor with different phosphorus status. Biol. Fertil. Soils 27:197–204. doi:10.1007/s003740050420

Sahrawat, K.L., D.R. Keeney, and S.S. Adams. 1987. Ability of nitrapyrin, dicyandiamide and acetylene to retard nitrification in a mineral and organic soil. Plant Soil 101:179–182. doi:10.1007/BF02370642

Shi, Y., D.B. Parker, N.A. Cole, B.W. Auvermann, and J.E. Melhorn. 2001. Soil amendments to minimize ammonia emissions from beef cattle feedlots. Trans. ASAE 44:677–682.

Singurindy, O., M. Molodovskaya, B.K. Richards, J. Warland, and T.S. Steenhuis. 2007. Emission of nitrous oxide from New York State dairy farms. Proceedings of the ASABE International Symposium on Air Quality and Waste Management for Agriculture, 16–19 Sept. 20007. ASABE, St. Joseph, MI. doi:10.13031/2013.23881

Simek, M., and J.E. Cooper. 2002. The influence of soil pH on denitrification: Progress towards the understanding of this interaction over the last 50 years. Eur. J. Soil Sci. 53:345–354. doi:10.1046/j.1365-2389.2002.00461.x

Smith, R.A., G.E. Schwarz, and R.B. Alexander. 1997. SPARROW surface water-quality modeling nutrients in watersheds of the conterminous United States: Model predictions for total nitrogen (TN) and total phosphorus (TP). USGS, Washington, DC. http://water.usgs.gov/nawqa/sparrow/wrr97/results.html (accessed 2 Mar. 2016).

Smith, L.C., C.A.M. de Klein, and W.D. Catto. 2008. Effect of dicyandiamide applied in a granular form on nitrous oxide emissions from a grazed dairy pasture in Southland, New Zealand. N. Z. J. Agric. Res. 51:387–396. doi:10.1080/00288230809510469

Smith, P., D. Martino, Z. Cai, D. Gwary, H. Janzen, P. Kumar, et al. 2007. Agriculture. In: B. Metz, O.R. Davidson, P.R. Bosch, R. Dave, and L.A. Meyer, editors, Climate change 2007: Mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge Univ. Press, Cambridge, UK and New York. p. 497–540.

Stackhouse, K.R., Y. Pan, Y. Zhao, and F.M. Mitloehner. 2011. Greenhouse gas and alcohol emissions from feedlot steers and calves. J. Environ. Qual. 40:899–906. doi:10.2134/jeq2010.0354

Stackhouse-Lawson, K.R., M.S. Calvo, S.E. Place, T.L. Armitage, Y. Pan, Y. Zhao, et al. 2013. Growth promoting technologies reduce greenhouse gas, alcohol, and ammonia emissions from feedlot cattle. J. Anim. Sci. 91:5438–5447. doi:10.2527/jas.2011-4885

Subbarao, G.V., O. Ito, K.L. Sahrawat, W.L. Berry, K. Nakahara, T. Ishikawa, et al. 2006. Scope and strategies for regulation of nitrification in agricultural systems- Challenges and opportunities. Crit. Rev. Plant Sci. 25:303–335. doi:10.1080/07352680600794232

Suwa, Y., Y. Imamura, T. Suzuki, T. Tashiro, and Y. Urushigawa. 1994. Ammonia-oxidizing bacteria with different sensitivities to $(NH_4)_2SO_4$ in activated sludges. Water Res. 28:1523–1532. doi:10.1016/0043-1354(94)90218-6

Taylor, A.E., L.H. Zeglin, S. Dooley, D.D. Myrold, and P.J. Bottomley. 2010. Evidence for different contributions of archaea and bacteria to the ammonia-oxidizing potential of diverse Oregon soils. Appl. Environ. Microbiol. 76:7691–7698. doi:10.1128/AEM.01324-10

Todd, R.W., N.A. Cole, M.B. Rhoades, D.B. Parker, and K.D. Casey. 2011. Daily, monthly, seasonal, and annual ammonia emissions from southern High Plains cattle feedyards. J. Environ. Qual. 40:1090–1095. doi:10.2134/jeq2010.0307

Tsutsui, H., T. Fujiwara, K. Matsukawa, and N. Funamizu. 2013. Nitrous oxide emission mechanisms during intermittently aerated composting of cattle manure. Bioresour. Technol. 141:205–211. doi:10.1016/j.biortech.2013.02.071

Upadhyay, L.S.B. 2012. Urease inhibitors: A review. Indian J. Biotechnol. 11:381–388.

USDA–ERS. 2012. Cattle and beef. USDA Economic Research Service, Washington, DC. http://www.ers.usda.gov/topics/animal-products/cattle-beef/background.aspx (accessed 1 Feb. 2016).

USDA–NASS. 2007. Census of agriculture: Summary and state data. USDA–NASS, Washington, DC. https://agcensus.usda.gov/Publications/2007/ (accessed 1 Aug. 2016).

USDA–NASS. 2013. Cattle on feed. USDA–NASS, Washington, DC. http://www.nass.usda.gov/Statistics_by_Subject/index.php?sector=ANIMALS & PRODUCTS (accessed 19 Jan. 2014).

USDA–NASS. 2015. Cattle industry: 2012 census of agriculture highlights. USDA–NASS, Washington, DC. http://www.agcensus.usda.gov/Publications/2012/Online_Resources/Highlights/Cattle/Cattle_Highlights.pdf (accessed 6 Apr. 2016).

USDA–NRCS. 2002. Conservation innovation grants. USDA–NRCS, Washington, DC. http://www.nrcs.usda.gov/wps/portal/nrcs/main/national/programs/financial/cig/ (accessed 8 Sept. 2016).

USEPA. 1990. Animal feeding operations (AFOs). National Pollutant Discharge Elimination System, USEPA, Washington, DC. https://www.epa.gov/npdes/animal-feeding-operations-afos (accessed 8 Sept. 2016).

USEPA. 2002. Federal water pollution control act [as amended through P.L. 107-301, 27 Nov. 2002]. USEPA, Washington, DC. http://www.epw.senate.gov/water.pdf (accessed 8 Sept. 2016).

USEPA. 2009. Technical support document for manure management systems: Proposed rule for mandatory reporting of greenhouse gases. Climate Change Division, Office of Atmospheric Programs, USEPA, Washington, DC. http://www2.epa.gov/sites/production/files/2015-07/documents/tsd_manure_020409.pdf (accessed 15 Jan. 2016).

USEPA. 2013. Inventory of U.S. greenhouse gas emissions and sinks: 1990-2011. USEPA, Washington, DC. http://epa.gov/climatechange/Downloads/ghgemissions/US-GHG-Inventory-2013-Main-Text.pdf (accessed 17 Jan. 2014).

USEPA. 2015. Overview of greenhouse gases: Nitrous oxide emissions. USEPA, Washington, DC. http://www3.epa.gov/climatechange/ghgemissions/gases/n2o.html (accessed 1 Feb. 2016).

Venterea, R.T. 2010. Simplified method for quantifying theoretical underestimation of chamber-based trace gas fluxes. J. Environ. Qual. 39:126–135. doi:10.2134/jeq2009.0231

Venterea, R.T. 2013. Theoretical comparison of advanced methods for calculating nitrous oxide fluxes using non-steady state chambers. Soil Sci. Soc. Am. J. 77:709–720. doi:10.2136/sssaj2013.01.0010

Waldrip, H.M., N.A. Cole, and R.W. Todd. 2015a. Nitrogen sustainability and beef cattle feedyards: I. Introduction and influence of pen surface conditions and diet. Prof. Anim. Sci. 31:395–411. doi:10.15232/pas.2014-01361

Waldrip, H.M., N.A. Cole, and R.W. Todd. 2015b. Nitrogen sustainability and beef cattle feedyards: II. Ammonia emissions. Prof. Anim. Sci. 31:89–100. doi:10.15232/pas.2015-01395

Waldrip, H.M., K.D. Casey, R.W. Todd, and N.A. Cole. 2015c. Predicting greenhouse gas emissions from beef cattle feedyard manure. 2015 ASA-SSSA-CSSA International Meeting, Minneapolis, MN. 15–18 Nov. 2015. ASA-SSSA-CSSA, Madison, WI.

Waldrip, H.M., K.D. Casey, R.W. Todd, and N.A. Cole. 2015d. Factors affecting greenhouse gas emissions from beef cattle feedyard manure: A laboratory study. 2015 ASA-SSSA-CSSA International Meeting, Minneapolis, MN. 15–18 Nov. 2015. ASA-SSSA-CSSA, Madison, WI.

Ward, G.M., T.V. Muscato, D.A. Hill, and R.W. Hansen. 1978. Chemical composition of feedlot manure. J. Environ. Qual. 7:159–164. doi:10.2134/jeq1978.00472425000700020001x

Wijler, J., and C.C. Delwiche. 1954. Investigations on the denitrifying processes in soil. Plant Soil 5:155–169. doi:10.1007/BF01343848

Wileman, B.W., D.U. Thomson, C.D. Reinhardt, and D.G. Renter. 2009. Analysis of modern technologies commonly used in beef cattle production: Conventional beef production versus nonconventional production using meta-analysis. J. Anim. Sci. 87:3418–3426. doi:10.2527/jas.2009-1778

Willers, H.C., P.J.W. ten Have, P.J.L. Derikx, and M.W. Arts. 1993. Temperature-dependency of nitrification and required anoxic volume for denitrification in the biological treatment of veal calf manure. Bioresour. Technol. 43:47–52. doi:10.1016/0960-8524(93)90081-L

Williams, M., T.L. Richard, and E.F. Wheeler. 2008. Compaction of bedded dairy manure storage to reduce ammonia and greenhouse gas emissions. Proceedings of the 2008 ASABE Annual International Meeting, Providence, RI. 29 June–2 July 2008. ASABE, St. Joseph, MI. p. 7392–7402.

Woodbury, B.L., D.N. Miller, J.A. Nienaber, and R.A. Eigenberg. 2001. Seasonal and spatial variations of denitrifying enzyme activity in feedlot soil. Trans. ASAE 44:1635–1642. doi:10.13031/2013.7050

Yamada, T., S. Araki, W. Ikeda-Ohtsubo, K. Okamura, A. Hiraishi, H. Ueda, et al. 2013. Community structure and population dynamics of ammonia oxidizers in composting processes of ammonia-rich livestock waste. Syst. Appl. Microbiol. 36:359–367. doi:10.1016/j.syapm.2013.02.001

Yao, Y., B. Gao, M. Zhang, M. Inyang, and A.R. Zimmerman. 2012. Effect of biochar amendment on sorption and leaching of nitrate, ammonium, and phosphate in a sandy soil. Chemosphere 89:1467–1471. doi:10.1016/j.chemosphere.2012.06.002

Zaman, M., and M.L. Nguyen. 2012. How application timings of urease and nitrification inhibitors affect N losses from urine patches in pastoral system. Agric. Ecosyst. Environ. 156:37–48. doi:10.1016/j.agee.2012.04.025

Zhu, G., M. Ma, Z. Gao, W. Ma, J. Li, and Z. Cai. 2014. Characterizing CH_4 and N_2O emissions from an intensive dairy operation in summer and fall in China. Atmos. Environ. 83:245–253. doi:10.1016/j.atmosenv.2013.11.001

Zhu-Barker, X., A.R. Cavazos, N.E. Ostrom, W.R. Howath, and J.B. Glass. 2015. The importance of abiotic reactions for nitrous oxide production. Biogeochemistry 126:251–267. doi:10.1007/s10533-015-0166-4

Zumft, W.G. 1997. Cell biology and molecular basis of denitrification. Microbiol. Mol. Biol. Rev. 61:533–616.