frontiers | Frontiers in Animal Science

TYPE Review
PUBLISHED 02 July 2025
DOI 10.3389/fanim.2025.1608387

OPEN ACCESS

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RECEIVED 08 April 2025
ACCEPTED 03 June 2025
PUBLISHED 02 July 2025

CITATION
Lee M, Auvermann BW, Tedeschi LO, Koziel JA, Brandani CB, Gouvêa VN, Smith JK and Casey KD (2025) Ammonia emissions from beef cattle feedyards: a review. *Front. Anim. Sci.* 6:1608387. doi: 10.3389/fanim.2025.1608387

Ammonia emissions from beef cattle feedyards: a review

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This review described the state of the science concerning the generation, measurement, and mitigation of ammonia (NH_3) emissions from beef cattle feedyards. NH_3 emissions primarily come from urinary urea in cattle manure. In the past, constant emission factors were used to inventory NH_3 emissions. Currently, NH_3 emission factors estimated by process-based mechanistic models reflecting various factors affecting NH_3 emissions in the feedyard environment are available. This review of current literature indicated the average NH_3 emissions from a beef cattle feedyard was approximately 119 g/head/day (range 24 to 318 g/head/day), and the average NH_3 flux was approximately 58 μg/m^2/s (range 2 to 185 μg/m^2/s). Although more realistic estimates of NH_3 emission flux from open-lot livestock facilities were being obtained using process-based models, there was still significant variation depending on the diet composition, manure management practices, and the feedyard environment, including both seasonal weather patterns and synoptic weather events. We note the need to improve inventories of NH_3 emissions into categories of crude protein percentage, manure management implemented, and feedyard environment. Some mitigation strategies can be effective, such as diet manipulation, growth-promoting technologies, and manure or pen-surface amendments. Of those, precision diet feeding to meet but not exceed protein requirements appeared to be the most practical way to reduce ammonia emissions over the animals' feeding period; laboratory studies suggested that shorter-term reductions in emission flux may be possible with the other approaches, but they were far more speculative at this point as to both their efficacy and their cost of implementation.

KEYWORDS

gas quantification, emission factors, emission mitigation, feedyard management practices, air quality, sustainable agriculture

1 Introduction

Over the past few decades, livestock and poultry farmers have scaled up farming operations to meet society's demand for high-quality meats, milk, eggs, and by-products. The concentration of animals in feeding operations has played a large role in fulfilling the demand for animal protein with fewer animals and while using fewer land resources. Concentration of animals in close proximity during a portion of their production cycle also concentrates their nutrient emissions. More specifically, these undesirable potential implications of confined (or "concentrated") animal feeding operations (CAFOs) are caused by gas and particulate matter (PM) emissions from various types of animal wastes, including manure (feces and urine), waste feed, bedding, and wastewater. Gaseous emissions from CAFOs include NH_3, greenhouse gases (carbon dioxide, CO_2; methane, CH_4; and nitrous oxide, N_2O), and other air pollutants such as volatile organic compounds (VOCs), many of which are odorous.

NH_3 emissions from CAFOs are a high-profile environmental quality concern because they can contribute to the eutrophication of surface waters, nitrate contamination of ground waters, soil acidity, secondary formation of fine PM, and impaired air quality (USEPA, 2004; Hribar, 2010; Brandani et al., 2023). Gaseous NH_3 in the atmosphere has been reported as a significant contribution to the formation of airborne fine particulate matter (PM2.5) through reactions with water vapor and other air pollutants, including oxidation products of sulfur dioxide (SO_2) or nitrogen oxides (NO and NO_2, or NOx) (Li et al., 2008; Wyer et al., 2022). Indeed, U.S. Environmental Protection Agency (USEPA) recently reduced the annual health-based National Ambient Air Quality Standard for PM2.5 from 12.0 $\mu g/m^3$ to 9.0 $\mu g/m^3$ (USEPA, 2024). Since NH_3 is a precursor gas that may be easier to mitigate than others among PM2.5's other precursors (Meng et al., 2017; Gu et al., 2021; Wyer et al., 2022), if ambient PM2.5 standards are further reduced, ambient air-quality standards for NH_3 may be introduced. Furthermore, NH_3 may also contribute to climate change through N_2O formation as an intermediate byproduct of ammonium (NH_4^+) oxidation in the microbial processes of nitrification and denitrification (USEPA, 2010). In addition, NH_3 emissions may contribute to nitrogen (N) deposition in neighboring ecosystems, which in turn may affect ecosystem function by promoting eutrophication, soil acidification, and disrupting biodiversity (Benedict et al., 2013; Thompson et al., 2015; Morris, 2016; Brandani et al., 2023).

As public awareness and concern over the potential adverse effects of NH_3 emissions on the environment and human health increased, governmental regulation of CAFOs led to a push and the adoption of sustainable management practices by livestock producers (Waldrip et al., 2015). Sound scientific evidence is needed to evaluate current and proposed regulations that go beyond encouraging practices regarding CAFOs, air quality, PM, and NH_3 emissions in particular. It is necessary to understand the emission mechanisms and processes influencing emissions, know the appropriate measurement methodologies and techniques and uncertainties associated with their use, evaluate current scientific literature for feedyard-based emissions data, survey the industry feedyard management practices, anticipate the impact of emerging regulatory trends and socioeconomics on emissions, and identify the most practical approaches to mitigation and potential barriers to their adoption.

In this review, we reported the state of the science concerning NH_3 emissions from beef cattle feedyards. The review is organized into five major areas: 1) pathway of NH_3 emissions through N metabolism in the ruminant animal, 2) dynamics of NH_3 emissions from pen surfaces, 3) methods quantifying NH_3 emissions in the feedyard, 4) the current level of NH_3 emissions in beef cattle feedyards, and 5) recommended management practices to mitigate NH_3 emissions from feedyards. Details of the literature search methodology, including search engines, terms used, inclusion and exclusion criteria, and the total number of materials reviewed, are provided in the Supplementary Material.

2 Pathway of ammonia emissions from ruminant animal

Ruminants, specifically pre-gastric fermenters, have a unique digestive system that evolved to digest and use forage resources that are less or not digestible in monogastric animals. The unique digestive organ called the reticulo-rumen is an anaerobic microbe fermenter with the remarkable ability to convert dietary protein into microbial protein. Ruminal microorganisms can use not only dietary protein but also non-protein-N, which does not contain amino acids (e.g., urea and NH_3) as N sources for microbial protein synthesis. In the reticulo-rumen, N sources are degraded by rumen microbes to peptides, amino acids, and eventually to NH_3 by the deamination of amino acids and then these compounds are used to synthesize microbial protein (Hristov and Jouany, 2005; NASEM, 2016). Microbial protein has a similar amino acid composition to the amino acid composition of tissue and milk protein, which makes it almost an ideal source of amino acids for the ruminant (NRC, 2001; Hristov et al., 2011). The metabolizable protein needs of the ruminant for maintenance and production are met primarily by microbial proteins that are washed out of the reticulo-rumen and feed proteins, which are not degraded by rumen microbes, with a small contribution from endogenous N, which originates from the animal's own viscera rather than from dietary sources such as sloughed-off intestinal cells in the post-ruminal (small and large intestine) metabolism (NRC, 2001; NASEM, 2016). A portion of NH_3 produced in ruminal N metabolism is absorbed across the ruminal epithelium into the portal vein and converted mostly into urea by the urea cycle in the liver to avoid NH_3 toxicity (NASEM, 2016). Urea produced by the liver is partly excreted in the urine by the kidneys, with the remainder recycled back to the gastrointestinal tract (GIT) through either direct transfer from blood across the epithelial tissue or via saliva as a N source for protein synthesis (NASEM, 2016). The process in which NH_3 in the rumen is converted into urea by the liver and reused as a N source in GIT is called urea recycling (or N recycling), and it plays an important role in N preservation mechanism of the ruminant (NASEM, 2016).

Undigestible and unabsorbed N sources in ruminal and post-ruminal metabolism are excreted as feces. The overall N metabolism pathway in the ruminant is shown in Figure 1.

The excretion of NH_3 is largely determined by the form of NH_3 in the ruminant's metabolism. For example, if it is in a gas phase (NH_3), it is most likely to be excreted as eructation, exhaled gas, and flatus. However, if it is in an ionized (NH_4^+) or solid form, it is more likely to be incorporated into the manure (feces and urine). Since the form of NH_3 produced in the digestion of the ruminant is determined by the NH_3/NH_4^+ equilibrium state (Equation 1), it is important to understand the factors that affect the NH_3/NH_4^+ equilibrium and the ruminal and post-ruminal environments. The equilibrium between NH_3 and NH_4^+ is not a redox-dependent reaction, but a pH and temperature (T) dependent reaction in aqueous solutions (Equations 2, 3; Emerson et al., 1975) as illustrated in Figure 2. This is because there is no change in the oxidation states of N or hydrogen (H), which is the key feature of redox reactions.

$$NH_3 + H^+ \leftrightarrow NH_4^+ \tag{1}$$

$$\text{Mole fraction of } NH_3 = 1/[10^{(\text{pKa} - \text{pH})} + 1] \tag{2}$$

$$\text{pKa} = 0.0901821 + [2729.92/T(K)] \tag{3}$$

The ruminal and post-ruminal environments depend on diet composition, management, and cattle's health condition, but they are generally anaerobic, reductive (oxidation-reduction potential (Eh) of -250 to -450 mv; Van Soest, 1994),< 8 pH, and ~39°C (NASEM, 2016). Considering the general pH and T in the rumen and post-rumen, it is reasonably assumed that NH_3 exists in mostly the NH_4^+ form. In addition, Mohiuddin and Khattar (2019)

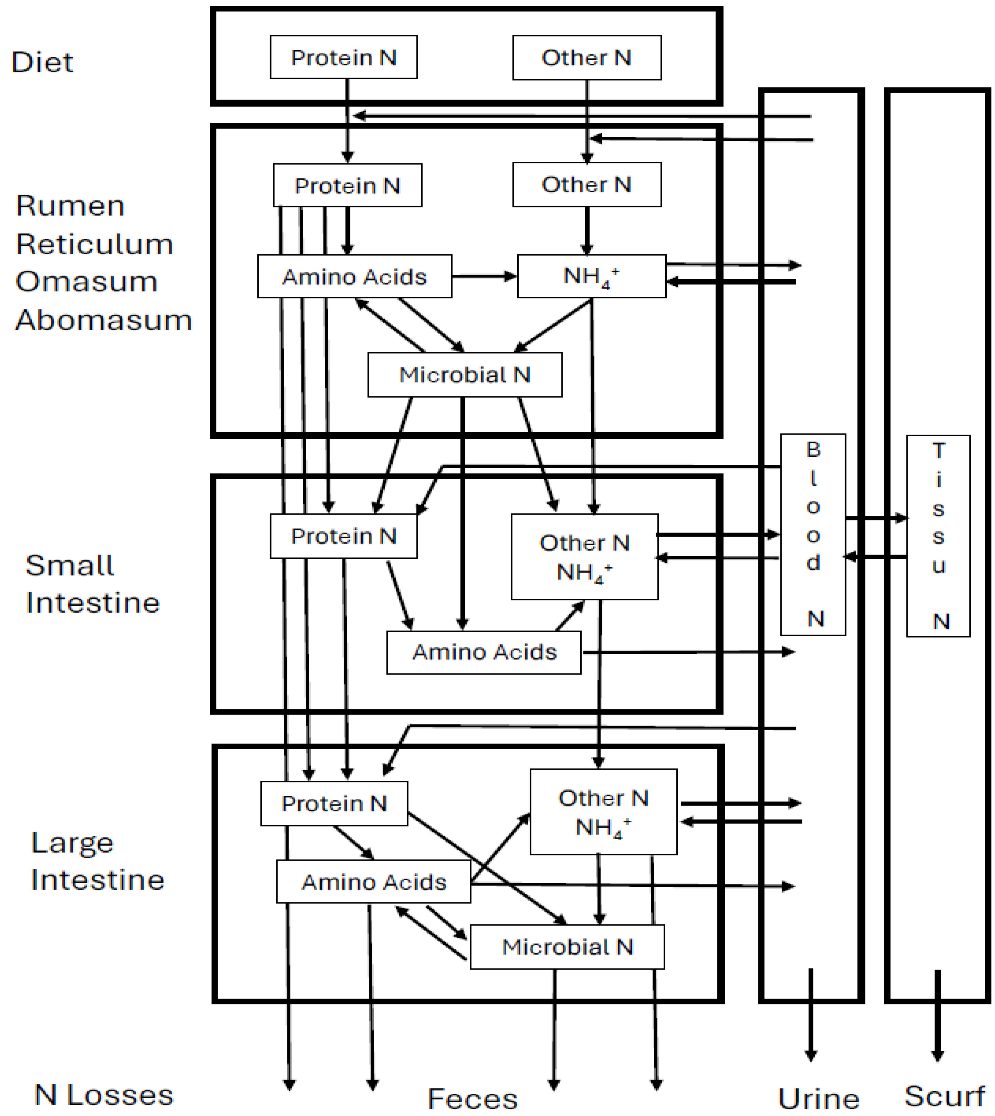

FIGURE 1
A schematic diagram of nitrogen metabolism in the ruminant. The diagram was reprinted from NRC (1985) and modified to incorporate concepts described by Tedeschi and Fox (2020).

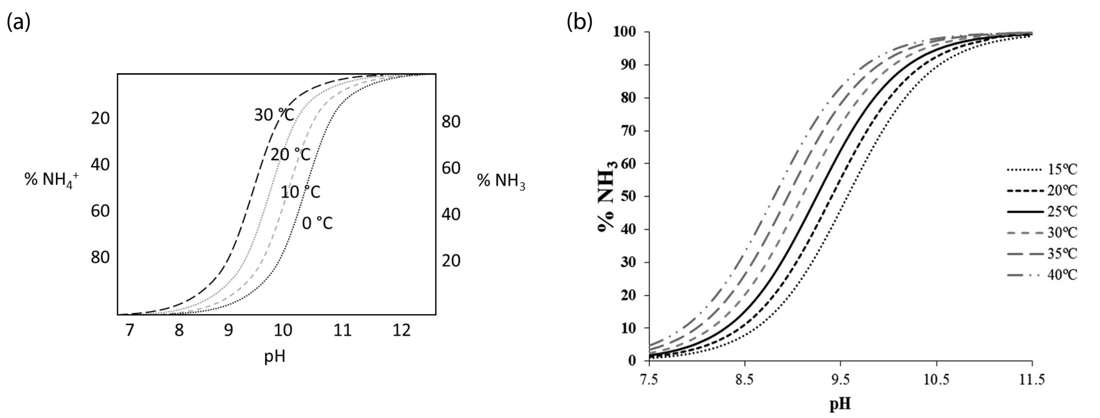

FIGURE 2
Equilibrium between NH_4^+ and NH_3 as a function of pH and temperature. The images were reprinted from **(a)** Cofie et al. (2016), **(b)** Aguado et al. (2022).

reported that the pKa of this reaction is about 9.15 and this reaction toward NH_4^+ occurs almost instantaneously under biological conditions (pH 7.4 and 36.5°C). However, the aqueous solution in the rumen and post-rumen is more dynamic and complex due to anaerobic microbial interactions and reductive conditions, thus in addition to pH and T, the change in pressure, ionic strength, and salinity may affect the conversion of NH_3 form due to byproducts from microbial digestion and GIT metabolism.

Thus, based on the data investigated to date, we aimed to discuss specifically the form of NH_3 estimated to be emitted by each NH_3 emission pathway: 1) exhalation, 2) eructation, 3) flatus, and 4) the excreted N sources in manure, considering the unique N metabolism and the viscera environment of the ruminant. Environmental conditions and the predominant forms of NH_3 associated with each NH_3 emission pathway are described in the Supplementary Material. In summary, since pH and T are widely recognized as the primary factors influencing the chemical form of NH_3 under biological conditions, NH_4^+ is the dominant form in the ruminant's GIT. Consequently, most NH_3 emitted from ruminants is considered to originate from excreted N in manure (Figure 3).

3 Dynamics of ammonia emissions from pen surfaces

Excreted N in feces and urine is the main source of NH_3 emissions from the beef cattle feedyard. Fecal NH_3 is derived from undigested feed residues, microbial cells, endogenous secretions, sloughed cells from GIT (Waldrip et al., 2015), and urine NH_3 derived from urea, hippuric acid, and purine-based catabolism residues (Bristow et al., 1992).

Urea ($CO(NH_2)_2$) is not volatile, but once it comes in contact with the urease enzyme (urea amidohydrolase), which is ubiquitous in manure and soil (Waldrip et al., 2015), it is rapidly hydrolyzed to NH_3 and CO_2 (Bussink and Oenema, 1998). However, NH_3 from feces is generated through the slow process of organic N mineralization (Muck, 1982; Muck and Steenhuis, 1982; Waldrip et al., 2015). N compounds from feces are mineralized into NH_4^+ by heterotrophic microorganisms in the manure (Horton et al., 2006; Zhang et al., 2007; Vavilin et al., 2008). Mineralized NH_4^+ is slowly released as NH_3 by diffusive and convective mass transfer (Waldrip et al., 2015). The process for volatilizing NH_3 from cattle manure was summarized below (Figure 4, Equations 4-6; Brandani et al., 2023).

$$CO(NH_2)_2 \ + \ 2H_2O \ + Urease \ \rightarrow \ (NH_4)_2CO_3 \tag{4}$$

$$(NH_4)_2CO_3 \ + 2H^+ \ \rightarrow 2NH_4^+ \ + \ CO_2 \ + \ H_2O \tag{5}$$

$$NH_4^+ \cdot OH^- \ \leftrightarrow \ NH_3 \ + \ H_2O \tag{6}$$

Most NH_3 emission from the beef feedyard originates from urine NH_3, particularly urinary urea (Bristow et al., 1992; Bussink and Oenema, 1998; Waldrip et al., 2013a). The urinary N could be volatilized from 4% to 71% (Bussink and Oenema, 1998; Waldrip et al., 2013a), while feces N volatilization is considerably less at 1% to 13% (Bussink and Oenema, 1998). Supporting this, Lee and Hristov (2010) observed that urinary N accounted for an average of 90% of the total emitted NH_3 during the first 10 d after excretion as cattle manure. In addition, several studies have reported that approximately 80% (range: 25 to 90%) of the urinary N is volatilized to NH_3 within the first 24 h after manure excretion (Stewart, 1970; James et al., 1999; Cole and Todd, 2009; Lee et al., 2009). Urea represents 50% to 90% or more of total urinary N (Bussink and Oenema, 1998; Reynal and Broderick, 2005; Vander Pol et al., 2008) and proportionally increases as dietary CP level and intake increase (Cole et al., 2005; Colmenero and Broderick, 2006; Todd et al., 2006; Cole and Todd, 2009; Waldrip et al., 2013a). Waldrip et al. (2013a) reported a moderate relationship ($R^2 = 0.516$) between dietary CP % and urinary N in finishing beef cattle.

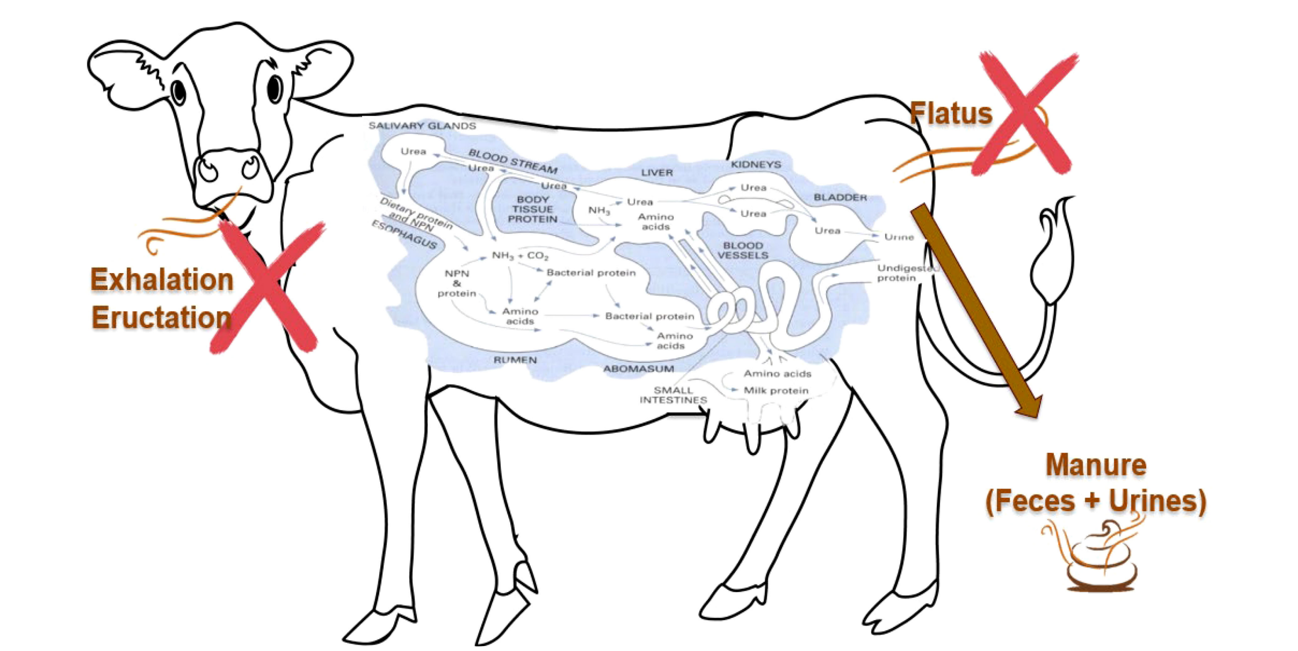

FIGURE 3
A schematic of ammonia emission pathways from the ruminant. The protein pathways inside the ruminant were reprinted from Vaga (2017).

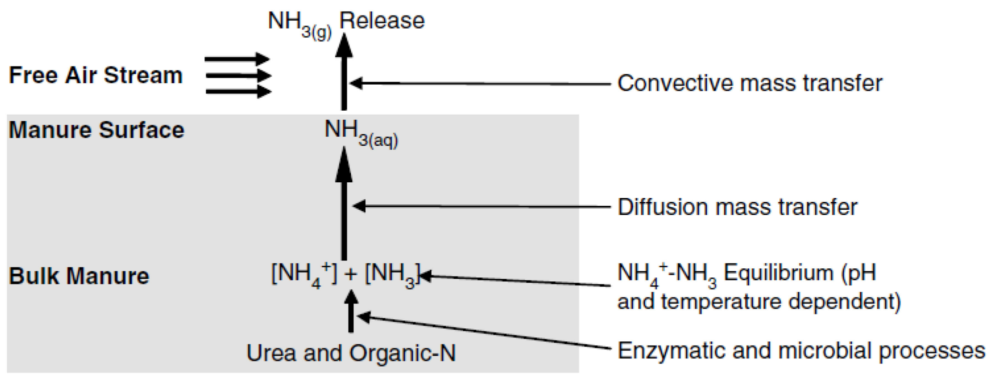

FIGURE 4
A conceptual model of NH_3 formation and volatilization. The image was reprinted from Hristov et al. (2011).

The instantaneous magnitude and rate of NH_3 loss are the result of complex physical and chemical processes on feedyard surfaces (Harper, 2005; Freney and Simpson, 2013). They depend strongly on diet composition, manure properties, environmental factors (T, precipitation, humidity, and wind turbulence), manure properties, and management practices (Sommer et al., 1991; Ni, 1999; Brandani et al., 2023). In summary, NH_3 emission is mostly driven by four factors (Harper et al., 2010a): 1) total ammonia N (TAN) concentration of the manure, 2) T of the manure and pen surfaces, 3) pH of the manure and pen surfaces, and 4) the effectiveness of mass transfer and turbulent transport of the NH_3 away from the manure surface. The relationship between NH_3 volatilization and four key factors was summarized in the Supplementary Material. In conclusion, NH_3 volatilization increases with increasing TAN concentration, T, wind speed, and pH (Sawyer et al., 1978; Sommer et al., 1991; Arogo et al., 2006; Montes et al., 2009; Todd et al., 2011). Temperature and pH have been reported to be the most important factors influencing NH_3 volatilization (Arogo et al., 2006), as NH_3/NH_4^+ are equilibrium-dependent (Figure 4). Supporting this, a significant correlation between the above parameters and NH_3 volatilization was reported in Redding et al. (2019) (Table 1).

4 Quantifying ammonia emissions

To accurately quantify NH_3 emissions in the feedyard, it is necessary to understand the characteristics of NH_3 emissions

TABLE 1 Correlation coefficient observed between environmental factors and NH_3 volatilization.

Environmental factor	Correlation coefficient (Kendall s tau-b)	*p*-value
Wind friction velocity	0.34	<0.01
Manure temperature	0.36	<0.05
Air temperature	0.16	<0.05
Temperature difference between the manure and the air	0.46	<0.05
Cattle numbers in the feedyard	0.21	<0.01

occurring in the feedlot environment. NH_3 is a colorless gas with a distinct, pungent smell. It occurs naturally and is normally found in trace amounts in the atmosphere (range: 1 to 25 ppb; Renard et al., 2004). Due to its high reactivity and the pervasiveness of the urease enzyme, the process of NH_3 formation and volatilization is almost instantaneous and begins immediately after manure is excreted (Hristov et al., 2011). Once emitted into the atmosphere, NH_3, where it is the dominant alkaline gas, reacts with atmospheric sulfuric and nitric acids forming ammonium sulfate, ammonium bisulfate, and ammonium nitrate, which precipitate in atmospheric water droplets as secondary fine particles (PM2.5) and are regulated by USEPA as a so-called "criteria" air pollutant (Renard et al., 2004; Brandani et al., 2023). NH_3 released into the atmosphere has a short lifespan in its gas phase of 2.5 to 36 h (Xie et al., 2024).

NH_3 emissions from open feedyards are generally lower than those encountered in closed or housed animal production systems (Todd et al., 2005, 2006; Hristov et al., 2011). This is because open feedyards are exposed to ambient air, allowing for dilution airflow that reduces NH_3 concentrations in the atmospheric boundary layer. Although ambient conditions are spatially and temporally variable, NH_3 emissions from beef feedyards are quickly dispersed by atmospheric turbulence (Waldrip et al., 2015). In other words, most agricultural open sources like a feedyards tend to be scattered both temporally and spatially, and most of the gaseous NH_3 emitted by feedyards may be shortly adsorbed to surrounding cropping and natural ecosystems by dry deposition (Harper et al., 2004; Harper, 2005), converted to fine particles, or mixed into the upper atmosphere. Additionally, manure in open feedyards is typically distributed over a larger area and may dry out more quickly due to exposure to sunlight and wind. Although more NH_3 may be released during the drying process, dried manure emits less NH_3 compared to the wetter manure often found in closed systems. In addition, open-lot feedyards often have lower stocking density compared to animal operations under the roof, which reduces the total amount of NH_3 being produced per unit of emitting area.

Quantifying gas emissions from open sources requires equipment that can measure low concentrations of NH_3 quickly, accurately, and robustly. Any measurement procedure that alters the natural ambient state (e.g., manure property and turbulence at the emitting surface) will introduce bias to measured NH_3 emission rates (Harper et al., 2010a). For NH_3, any measurement technology that interferes with the turbulent transport process away from the source (the rate-limiting process) can result in large errors. This is unlike CH_4, CO_2, or NO_2, which are less soluble and less affected by turbulent transport as mass-flow (biological) gases (Harper et al., 2010a). Therefore, measuring the NH_3 concentration emitted in the natural ambient state with the acquisition of weather data is better for the quantification of NH_3 emission in the feedyard compared to other approaches such as creating an artificial airflow inside a flux chamber.

Quantifying NH_3 emissions requires at least two components: 1) a method to measure the atmospheric concentration of NH_3 and 2) a method to measure weather data for converting the concentration into emission using a dispersion model or a method to directly measure the airflow rate (usually for lab scale) (Waldrip et al., 2015). It is important to note that concentration is only a percentage. 1 ppm of NH_3 just means that NH_3 is 0.0001% of the sampled air. It says nothing about the actual amount of NH_3 injected into the atmosphere. Therefore, it is necessary to approach the term of emissions (mass per time). If the total volumetric airflow (m^3/s) and the NH_3 concentration (g/m^3) from an emission source were measured, the two terms must be multiplied to obtain the emission rate in g/s. The main text of this review describes only the most widely used method currently applied in practice. However, the Supplementary Material provides an overview of five major approaches for measuring NH_3 concentrations (acid trap, chemiluminescence, electrochemical sensor, infrared analyzer, and tunable diode analyzer) and estimating NH_3 emissions (N mass balance, flow-through chamber, micrometeorological methods, air dispersion models, and satellite remote sensing). As each method has advantages, disadvantages, and applicability varies, readers are encouraged to consult all methods and select the one most appropriate for their specific research conditions and objectives.

4.1 Measurement of ammonia concentration

4.1.1 Open-path tunable diode laser absorption spectrometry

Open-path tunable diode laser absorption spectrometry (OP-TDLAS) is a technique designed to measure the path-averaged concentration of specific species within a gas mixture using laser-absorption spectrometry. The basic principle of OP-TDLAS involves passing the laser through the gas mixture, detecting the amount of light absorbed by NH_3 molecules at specific wavelengths according to the change in the degree of recovery rate from the detector, and then computing the NH_3 concentration based on the calibration between NH_3 concentration and the amount of light absorbed at certain wavelengths (Figure 5). Such a specific waveband (so-called narrow absorption line), specifically designed for NH_3, avoids mutual absorption interference of other gases such as CO_2, CH_4, and water vapor (Harper et al., 2010a). The amount of light absorbed by the NH_3 molecules is proportional to the concentration of NH_3 along the optical path. NH_3 gas molecules

FIGURE 5
A figure of the OP-TDLAS on a beef cattle feed yard.

typically absorb light in the range of around 200 (Boreal Laser INC), 620~740, or 931~954 cm^{-1} (Baldacchini et al., 1981; Hermanussen et al., 1986). To sum up, OP-TDLAS is designed to measure mean concentrations along an open path between the laser and the retroreflector and is a non-invasive technique.

The main advantage of using OP-TDLAS at beef cattle feedyards is quick, accurate, and robust NH_3 measurement in a feedyard environment where NH_3 rapidly volatilizes from relatively large emitting areas. As an example, the detailed specification of OP-TDLAS by Boreal Laser Inc (Edmonton, Canada) has 8–6500 or 40-15,000 ppm-m as a detectable NH_3 range and a ±2% uncertainty about reading accuracy. The response time required for measuring accurate NH_3 concentration is 1 s. Once factory calibration is completed, it has a longer calibration cycle than other sensors. If stored properly, the measurement will remain accurate for several years. In addition, the open path between the laser and retroreflector can typically be covered to 5~500 m in measurements, but this can be increased further depending on the performance of the reflector. In terms of disadvantages, OP-TDLAS is expensive and requires careful maintenance. It may require skilled operators for setup, calibration, and maintenance due to the complexity of the technology involved. It is susceptible to maintaining clear line of sight between the laser and retroreflector. Environmental conditions like dust and condensation can increase the opacity of the air along the optical path and degrade the quality of an instrument's signal.

4.2 Estimation of ammonia emissions

4.2.1 Air dispersion models

Direct measurement of NH_3 emissions from open cattle feedyards is challenging due to their size, the spatial and temporal variable nature of emissions from open sources, and the labor, cost, and time consumption associated with measuring and maintaining instruments (Bonifacio et al., 2013). One may sidestep these problems by using an atmospheric dispersion model to deduce the emission indirectly (Flesch et al., 2004). Air dispersion models are mathematical tools used to characterize the atmospheric processes that disperse a pollutant emitted by a source and simulate the transport and dispersion of air pollutants in the atmosphere based on measured weather data and gas emissions (or concentration). These models help assess the NH_3 emission at various locations downwind from a source, providing valuable information for air quality management, environmental impact assessments, and regulatory compliance.

In the case of open beef feedyards, several air modeling systems could be applied such as 1) Gaussian model-based AERMOD (American Meteorological Society/Environmental Protection Agency Regulatory Model) system and 2) backward Lagrangian stochastic (bLS) model-based WindTrax system, but bLS model-based WindTrax is generally applicable for beef cattle feedyards.

4.2.1.1 Gaussian model-based AERMOD system

A Gaussian dispersion model describes the transport of pollutants from a point source as a steady-state plume whose horizontal and vertical spread are modeled as Gaussian distributions whose parameters are specified by ensembles of weather variables affecting boundary-layer turbulence. The AERMOD System is a steady-state, Gaussian plume model that incorporates air dispersion based on planetary boundary layer turbulence structure and scaling concepts, including treatment of both surface and elevated sources, and both simple and complex terrain (USEPA, available online: https://www.epa.gov/scram/air-quality-dispersion-modeling). It is the preferred regulatory dispersion model of USEPA and is a free system used for emission estimation of target gases across various industries. An advantage of such Gaussian model-based system is that the plume dispersion parameters are based on theory and inputs are well characterized by experimental data (Arogo et al., 2006). However, this Gaussian assumption is not valid for all variables associated with the atmosphere for all time, scales, and dynamics (Goodliff et al., 2020). Specifically, Harper et al. (2011) pointed out that it is hard to expect such universality of Gaussian distribution since the atmosphere in the feedyard does not adhere to Gaussian assumptions. In other words, the shortcomings and limitations of Gaussian model arise from the many simplifying assumptions implicit in the mathematical solutions of these models (such as conditions of steady, uniform flow and homogenous turbulence), and the assumption of vertical Gaussian concentration distribution which is often not realized in the boundary layer (Arogo et al., 2006).

4.2.1.2 bLS model-based WindTrax

Lagrangian stochastic models (LS) describe the trajectories of tracer particles in turbulence from a statistical perspective of random velocity fields. They are considered by some authors the most natural and accurate means of calculating atmospheric transport (Wilson and Sawford, 1996). Flesch et al. (1995) developed a "backwards" variant of this type of model, otherwise known as the bLS dispersion model. The bLS dispersion model calculates an ensemble of particle trajectories that are distinguished by each passing through an observation point (Flesch et al., 1995). In other words, particles are released at the receptor location and travel backward in time to the source location in bLS; by contrast, in forward or standard LS, particles are released at the source and travel to the receptor location (Li and Du, 2020). Specifically, the bLS model tracks the movement of individual air parcels or particles as they disperse in the atmosphere based on measured weather data and gas concentration. When used in conjunction with OP-TDLAS measurements of pollutant concentrations, thousands of model trajectories are calculated upwind of the OP-TDLAS path for the prevailing wind conditions (Harper et al., 2010a). The important information relating the concentration to the emissions is the set of trajectory intersections with the ground (touchdowns), and the needed concentration-emission rate (C-Q) relationship is determined by those touchdowns according to Equations 7, 8 (Flesch et al., 2004).

$$Q = \frac{(C - C_b)}{(C/Q)_{sim}} \tag{7}$$

$$(C/Q)_{sim} = \frac{1}{N}\sum \frac{2}{W_0} \tag{8}$$

Where: Q: NH_3 emission rate ($kg/m^2/s$) from the area source of known configuration. C: NH_3 concentration (mg/m^3); Cb = the background NH_3 concentration (mg/m^3). (C/Q)sim: a model prediction of the ratio of concentration to the emission. N: the total number of (computational) particles released from the source. W_0: the vertical velocity at touchdown within the source (m^3/min per surface area; m^2).

The advantages of the bLS dispersion model are its ability to accurately represent wind features near the ground, their role in gas transport, (Harper et al., 2011) and to be faster and more flexible in calculating turbulent dispersion from surface area sources than "forward" models (Flesch et al., 1995). Therefore, it is a particularly good choice for calculating the relationship between gas concentration and emission rate for ground-level sources and for concentration observations taken near the source (Harper et al., 2011). However, it assumes that the atmospheric surface layer is homogeneous, that flow is stationary and that the source strength is spatially uniform (Flesch and Wilson, 2005), assumptions that may be challenged by the complexity of some CAFOs (Hristov et al., 2011). To date, there is much previous research on the bLS being used to calculate gas emissions from feedyards (Harper et al., 2004; McGinn et al., 2007; Van Haarlem et al., 2008). Flesch et al. (2004) reported that bLS diagnoses the strength of a small ground-level source with small bias (within 2%), however, Harper et al. (2010a) and Harper et al. (2010b) compared the bLS accuracies to tracer gas studies, showing a nominal bLS accuracy of 100 ± 10%.

bLS model-based WindTrax is an easy-to-use graphical interface designed for the assessment of turbulent transport on the micro-meteorological scale and for simulating short-range atmospheric dispersion (for horizontal distances within about 1 km of the source) using bLS models (Thunder Beach Scientific, available online: http://www.thunderbeachscientific.com/). This program is free, and guidelines and introductions are provided so that users can use them correctly. Before running the program, users should carefully review the associated documentation for detailed guidance on model inputs, options, and best practices.

5 Ammonia emission factors from beef cattle feedyard

Although measurements of NH_3 emission have been improved, direct measurements at each feedyard are not feasible due to the time, cost, and labor required. In addition, NH_3 emissions in beef cattle feedyards vary greatly depending on the diet (e.g., CP%), environmental conditions (e.g., air T, wind speed, turbulence, and precipitation), and operation-specific management practices (e.g., stocking density, manure storage, feeding management, and manure handling). Thus, measurements taken at one point in time on one feedyard may not accurately capture seasonal and temporal fluxes of emissions that occur due to changes in weather, animal diet, or other management practices (Waldrip et al., 2015). Although the emissions cannot be represented in a single value due to variables in the operation-specific management practices and environment, the need for a standard representing NH_3 emissions in general from livestock operations for inventory purposes is being highlighted.

In the past, researchers focused on measuring emissions and comparing them to constant emission factors (EF), which are derived from the literature by selecting data from studies that measured emissions from operations that are assumed to represent production facilities for a specific livestock type and region. Although they are not a perfect standard, constant EF are often used by regulatory agencies and environmental advocacy groups to estimate the footprint of specific animal-production systems (Battye et al., 1994; USEPA, 2004; Eggleston et al., 2006). A review by Faulkner and Shaw (2008) identified a wide range of constant EF for NH_3 from beef cattle and proposed an annual NH_3 EF of 13.0 kg/animal for beef cattle feedyards, which is the same as that used by the USEPA for inventory purposes (USEPA, 2004). Despite frequent use in setting policy and inventory of emissions, constant EF have proven insufficient for quantifying gas fluxes from many systems, including feedyards (Todd et al., 2013; Waldrip et al., 2013b, 2014). This is because using a single EF and applying it universally cannot account for the previously discussed temporal and spatial differences in management practices and climatic conditions (NRC, 2002, 2003). Therefore, NRC (2003) identified the need for improved resolution in emissions reporting for animal agriculture and recommended a process-based modeling approach that includes mass balance constraints.

Process-based modeling uses mathematical models to simulate the many processes and interactions that occur within a system such as a beef cattle feedyard. Dynamic process-based models that quantify emissions based on classical principles of thermodynamics and kinetics potentially provide a cost-effective method of estimating emissions and evaluating how changing climate and management practices affect emissions from animal agriculture (Waldrip et al., 2015). Representative process-based modeling used to quantify NH_3 emissions from beef cattle feedyards includes IHF, modified mass difference approach, flux-gradient technique (ECV), and IDM.

We have summarized results reported to date for NH_3 flux (μg/m^2/s) and the NH_3 EF (per capita emission rates; PCER, g/head/d) using several process-based models by season (Tables 2, 3). Open

TABLE 2 Seasonal NH_3 flux from beef cattle feedyards.

Reference	Location	Measurement or Estimation Method		NH_3 flux (μg/m^2/s)				
		NH_3 concentration	NH_3 emission	Spring	Summer	Autumn	Winter	Annual
Hutchinson et al. (1982)	Colorado	Acid trap	Vertical gradient flux model	29	44			
Shi et al. (2001)	Texas	Acid trap	Flux chamber					55
Koziel et al. (2004)	Texas	Chemiluminescence	Flux chamber		28		5	
Todd et al. (2005)	Texas	Acid trap	Flux gradient model		70		34	36
Baek et al. (2006)	Texas	Chemiluminescence	Vertical gradient flux model		61		5	
Todd et al. (2006)	Texas	Acid trap	Integrated horizontal flux model	3	16	2	2	
Todd et al. (2007)	Texas	Acid trap or Chemiluminescence	Flux gradient model		72		39	
McGinn et al. (2007)	Alberta, Canada	Open path laser	bLS Inverse dispersion model		84			
Rhoades et al. (2008)	Texas	Chemiluminescence	bLS Inverse dispersion model	89	77			
Van Haarlem et al. (2008)	Alberta, Canada	Open path laser	bLS Inverse dispersion model			10		
Staebler et al. (2009)	Alberta, Canada	Open path laser	bLS Inverse dispersion model			76		
Cole and Todd (2009)	Texas	N mass balance			64		36	
Rhoades et al. (2010)	Texas	Chemiluminescence	bLS Inverse dispersion model	84	77	63	58	71
Galles et al. (2011)	Colorado	Acid trap	Flux chamer					83-109
Sun et al. (2015)	New Jersey	Open path laser	Eddy covariance				37	
Parker et al. (2016)	Texas	Chemiluminescence	Flux chamber	8-38				
McGinn et al. (2016)	Alberta, Canada	Open path laser	Inverse dispersion model					50
Shonkwiler and Ham (2018)	Colorado	Open path laser	bLS Inverse dispersion model		60			
	Colorado	Open path laser	FIDES Inverse dispersion model		48			
Wang et al. (2024)	Victoria Australia	Open path laser	bLS Inverse dispersion model	113	185	104	107	127
Range				**3-113**	**16-185**	**2~104**	**2-107**	**36-127**
Average				**52**	**68**	**57**	**36**	**76**
Total average								**58**

TABLE 3 Seasonal NH_3 emissions from beef cattle feedyards.

<table>
<tr><th rowspan="2">Reference</th><th rowspan="2">Location</th><th colspan="2">Measurement or Estimation Method</th><th colspan="5">NH_3 emission factors (g/head/day)</th></tr>
<tr><th>NH_3 concentration</th><th>NH_3 emission</th><th>Spring</th><th>Summer</th><th>Autumn</th><th>Winter</th><th>Annual</th></tr>
<tr><td>Hutchinson et al. (1982)</td><td>Colorado</td><td>Acid trap</td><td>Vertical gradient flux model</td><td colspan="2">50</td><td></td><td></td><td></td></tr>
<tr><td>Cole et al. (2006)</td><td>Texas</td><td colspan="2">N mass balance</td><td>108</td><td></td><td></td><td>66</td><td></td></tr>
<tr><td>McGinn et al. (2007)</td><td>Alberta, Canada</td><td>Open path laser</td><td>bLS Inverse dispersion model</td><td></td><td colspan="2">140</td><td></td><td></td></tr>
<tr><td>Flesch et al. (2007)</td><td>Texas</td><td>Open path laser</td><td>bLS Inverse dispersion model</td><td>151</td><td>149</td><td></td><td></td><td></td></tr>
<tr><td>Rhoades et al. (2008)</td><td>Texas</td><td>Chemiluminescence</td><td>bLS Inverse dispersion model</td><td>89</td><td>78</td><td></td><td></td><td></td></tr>
<tr><td>Todd et al. (2008)</td><td>Texas</td><td>Acid trap</td><td>bLS Inverse dispersion model</td><td>118</td><td>128</td><td></td><td>64</td><td></td></tr>
<tr><td>Van Haarlem et al. (2008)</td><td>Alberta, Canada</td><td>Open path laser</td><td>bLS Inverse dispersion model</td><td></td><td></td><td>318</td><td></td><td></td></tr>
<tr><td rowspan="2">Denmead et al. (2008)</td><td>Victoria, Australia</td><td>Chemiluminescence</td><td>bLS Inverse dispersion model</td><td></td><td>69</td><td></td><td></td><td></td></tr>
<tr><td>Queensland, Australia</td><td>Chemiluminescence</td><td>bLS Inverse dispersion model</td><td></td><td>24</td><td></td><td></td><td></td></tr>
<tr><td>Staebler et al. (2009)</td><td>Alberta, Canada</td><td>Open path laser</td><td>bLS Inverse dispersion model</td><td></td><td></td><td>245</td><td></td><td></td></tr>
<tr><td>Todd et al. (2009)</td><td>Texas</td><td colspan="2">N mass balance</td><td></td><td></td><td></td><td></td><td>82-149</td></tr>
<tr><td>Rhoades et al. (2010)</td><td>Texas</td><td>Chemiluminescence</td><td>bLS Inverse dispersion model</td><td></td><td></td><td></td><td></td><td>85</td></tr>
<tr><td rowspan="2">Todd et al. (2011)</td><td>Feedlot A, Texas</td><td>Open path laser</td><td>bLS Inverse dispersion model</td><td>110</td><td>158</td><td>122</td><td>71</td><td>115</td></tr>
<tr><td>Feedlot E, Texas</td><td>Open path laser</td><td>bLS Inverse dispersion model</td><td>73</td><td>103</td><td>83</td><td>60</td><td>80</td></tr>
<tr><td rowspan="2">Waldrip et al. (2013b)</td><td>Feedlot A, Texas</td><td>Manure-DNDC</td><td>bLS Inverse dispersion model</td><td></td><td></td><td></td><td></td><td>173</td></tr>
<tr><td>Feedlot E, Texas</td><td>Manure-DNDC</td><td>bLS Inverse dispersion model</td><td></td><td></td><td></td><td></td><td>77</td></tr>
<tr><td>Chen et al. (2015)</td><td>Victoria, Australia</td><td>Chemiluminescence</td><td>Integrated horizontal flux model</td><td></td><td></td><td></td><td>156</td><td></td></tr>
<tr><td>Sun et al. (2015)</td><td>New Jersey</td><td>Open path laser</td><td>Eddy covariance</td><td></td><td></td><td></td><td>63</td><td></td></tr>
<tr><td>Shen et al. (2016)</td><td>Victoria, Australia</td><td>Acid trap</td><td>bi-directional NH_3 exchange model</td><td colspan="2">126</td><td></td><td></td><td></td></tr>
<tr><td rowspan="2">McGinn and Flesch (2018)</td><td>Feedlot A, Alberta, Canada</td><td>Open path laser</td><td>Inverse dispersion model</td><td></td><td></td><td></td><td></td><td>117</td></tr>
<tr><td>Feedlot B, Alberta, Canada</td><td>Open path laser</td><td>Inverse dispersion model</td><td></td><td></td><td></td><td></td><td>100</td></tr>
<tr><td rowspan="2">Shonkwiler and Ham (2018)</td><td>Colorado</td><td>Open path laser</td><td>bLS Inverse dispersion model</td><td></td><td>89</td><td></td><td></td><td></td></tr>
<tr><td>Colorado</td><td>Open path laser</td><td>FIDES Inverse dispersion model</td><td></td><td>71</td><td></td><td></td><td></td></tr>
<tr><td>Redding et al. (2019)</td><td>Queensland, Australia</td><td>Laser absorption spectroscopy</td><td>bLS Inverse dispersion model</td><td colspan="2">127</td><td></td><td></td><td></td></tr>
</table>

(Continued)

TABLE 3 Continued

Reference	Location	Measurement or Estimation Method		NH_3 emission factors (g/head/day)				
		NH_3 concentration	NH_3 emission	Spring	Summer	Autumn	Winter	Annual
Golston et al. (2020)	Colorado	Open path infrared gas analyzer	Gaussian plume approach		55			
Wang et al. (2024)	Victoria Australia	Open path laser	bLS Inverse dispersion model	170	190	126	155	160
Range				**73~170**	**24~190**	**83~318**	**60~156**	**77~173**
Average				**112**	**104**	**172**	**91**	**114**
Total average								**119**

path laser was often used to measure the NH_3 concentration and bLS Inverse dispersion model was widely used to convert the measured NH_3 concentration into NH_3 emissions from the beef cattle feedyard. A wide range of NH_3 flux has been reported, from 2 to 185 μg/m^2/s (average 58 μg/m^2/s). It was generally found that NH_3 flux follows the following order: summer > autumn > spring > winter. Also, a wide range of NH_3 emissions have been reported ranging from 24 to 318 g/head/d, and average 119 g/head/d. The highest NH_3 emissions were observed in the autumn, but the variation was large in each season, thus, no significant differences were represented between seasons ($p > 0.05$). It was found that 9 to 116 (average 43) NH_3 kg/head/y are emitted annually.

Although it is agreed that more realistic NH_3 EF are being obtained using process-based models, there is still significant variation of estimated NH_3 EF depending on the diet, the model used, and the feedyard environment. This deviation is most likely caused by differences in geographical environment and management implemented in actual feedyards as well as inherent assumptions made in modeling. Therefore, when investigating EF in the future, a detailed description of the environment, diet, and management practices implemented by the feedyard is necessary to evaluate the impact of each category of manure management implemented and the feedyard environment and estimate accurate EF for each scenario. We suggest an investigation list of feedyard environments to aid in assessing NH_3 emissions in Table 4. The information on feedyard management and diet reported in the previous papers to date is insufficient to implement this categorization. Improved models and measurement equipment are still needed for estimating more accurate NH_3 EF measurements under the ever-changing feedyard environment in the future.

6 Management practices to mitigate ammonia emissions

The major best management practices (BMP) available for use to mitigate NH_3 emissions in open-lot livestock facilities have been listed and recommended by USDA-NRCS (Table 5; Brandani et al., 2023). BMP can be divided into management for pre- and post-excretion stages (Waldrip et al., 2015). The primary method of NH_3 mitigation in the pre-excretion stage is fundamentally precise diet feeding to meet beef cattle requirements, dietary manipulation to decrease N excretion (e.g., adjusting dietary CP concentration to meet animal's need through phase feeding or oscillating), and supplementation to increase animal production (e.g., use of growth-promoting technologies). Methods for the post-excretion stage include applying the manure amendments to suppress hydrolysis of excreted urinary urea (e.g., use of urease inhibitors) or to capture (or absorb) the generated NH_4^+ or NH_3 (e.g., use of biochar as absorbents).

The fundamental solution to mitigate NH_3 emission in beef cattle feedyards is to minimize N excretion in manure by optimizing pre-excretion stages management, but there is an intrinsic limitation in achieving this goal due to the inefficient utilization of feed N in the ruminant. The efficiency of N utilization in ruminants is typically low (around 25%) and highly variable (10% to 40%) compared with the higher efficiency of other production animals (Calsamiglia et al., 2010). The low efficiency implies that unutilized N is being released as manure into the environment. The excretion of manure with high N has the potential to increase NH_3 emissions into the environment. Any practice or condition that increases manure N content should generally be expected to increase NH_3 emissions. With typical finishing diets, approximately 10 to 20% of N intake is retained in animal tissues, 30 to 50% of fed N is excreted in the feces, and 40 to 70% of fed N is excreted in urine (Cole and Todd, 2009). Although it is most likely impossible to dramatically improve the inherently low efficiency of N utilization in cattle, the efficiency of N utilization can be improved through the understanding and modification of factors regulating the efficiency of N utilization in key processes, including N capture in the rumen, protein degradation, digestion and absorption in the GIT and AA utilization in peripheral tissues (Calsamiglia et al., 2010). In addition, proper processing of forages and feed, such as chopping and steam flaking, enhances digestibility, thereby improving feed efficiency. Therefore, the direction we should take to mitigate NH_3 emissions in feedyards is to maximize the use of feed N by optimizing the pre-excretion management while simultaneously minimizing the environmental impacts using post-excretion management. The summarized NH_3

TABLE 4 Investigation list of feedyard management and environment.

Section	Type	Items	Answer
Feed	Nutrient composition	DM (% as fed)	
		CP (% DM)	
		Starch (% DM)	
		NDF (% DM)	
		ADF (% DM)	
		TDN (g/kg DM)	
		NEg (Mcal/kg DM)	
	Use of specific feed ingredients to mitigate nitrogen emissions		
	Intake (kg of DM/head/day)		
	Average feeding time(s) and interval(s) (e.g., 0600 and 1400 hours, fed twice per day)		
Growth promotants and metabolic modifiers	Use of monensin, growth-promoting hormone implant, ß-adrenergic agonists, and/or others		
Animal	Initial body weight (kg)		
	Final body weight (kg)		
	Head count		
	Animal density (head/area)		
	Total days on feed		
Manure and pen surface management	Manure cleaning cycle (e.g., days or times per year)		
	Manure storage method (e.g., composting)		
	Use of manure amendments		
	Use of water sprinklers on the pen surface		
Weather	Precipitation events and/or severe wind events		
Activity records	Cattle receipt (date)		
	Cattle shipment (date)		
	Manure removal (date)		

DM, dry matter; CP, crude protein; NDF, neutral detergent fiber; ADF, acid detergent fiber; TDN, total digestible nutrients; NEg, net energy for gain.

TABLE 5 Best management practices (BMP) for open-lot livestock facilities to decrease NH_3 deposition.

Category	NRCS code	Management practices
Feed management	592	Diet manipulation Growth-promoting technologies Phase feeding
Manure amendment	632	Surface amendment and manure separation
Dust control	375	Water sprinkler
Pen maintenance	N/A	Manure harvesting and pen drainage

References from USDA-NRCS; conservation management practices.

mitigation practices are organized into the pre- and post-excretion stages below.

6.1 Ammonia mitigation practices in the pre-excretion stage

6.1.1 Precision feeding

The terminology of precision feeding was coined to suggest that livestock feeding can be fine-tuned to maintain or improve performance and better realize other benefits (Reddy and Krishna, 2009). In other words, the ingredients and chemical composition of the diet are modified over the growth stage of the animal so that the nutrient composition of the diet more closely meets the nutrient requirements of the animal and the excreted nutrients in manure are minimized (Brandani et al., 2023). The terminology of precision feeding has expanded its meaning to include mitigating the environmental impacts of animal production while maintaining or improving animal performance. The expansion of the term's meaning is fundamentally due to the improvement of nutritional models based on the accumulated knowledge of livestock nutrient requirements and use by the animal, along with the development of feeding systems, which have made it possible to feed livestock closer to their requirements, thus reducing wastes (such as wasted feed ingredient, water, manure, and gas emissions) while maintaining or improving animal performance. Details regarding the development of nutritional models, as well as opportunities for improvement in current nutrition models, are presented in the Supplementary Material.

Ideally, a nutritionist can balance animal performance with subsequent effects on the environment using current nutrient models. However, the question remains whether precision feeding can be realistically applied to a commercial beef cattle feedyard (Waldrip et al., 2015). This is because there are several challenges to overcome for precision feeding to be practical. Factors that limit the practicality of precision feeding include (1) variability in animal nutrient requirements, (2) seasonal and climatic effects, (3) variability in the composition of feed ingredients, (4) logistics, and (5) variability in the estimation of DMI (Cole, 2003). Most of these limitations revolve around the risk of adversely affecting animal health or performance and feedyard benefits (Waldrip et al., 2015). For this reason, many nutritionists often incorporate safety margins in their dietary formulations and feeding recommendations to protect against such factors in order to ensure that the diet meets the nutrient requirements of the animal. A practical example of such safety margins is formulating the diet to contain more CP% than is expected to be required to meet the animal's requirements. However, it is important to note that the decision to overfeed crude protein, as an example, may also be influenced by feed ingredient price and/or availability, such as when ingredient price encourages overfeeding N as a means of minimizing the cost of gain.

Although we acknowledge the practical limitations of implementing precision feeding, scientific advancements to date have led to the development of several effective strategies within precision feeding systems to mitigate NH_3 emissions. Representative examples include phase feeding and the use of growth-promoting technologies, which will be discussed in detail next. Dietary protein requirements decrease as cattle mature because of reduced protein deposition and simultaneous increase in fat deposition (Hristov et al., 2011; Waldrip et al., 2015). Phase feeding is a type of precision feeding where dietary protein concentrations are reduced late in the feeding period (Waldrip et al., 2015). Use of growth-promoting technologies are used in tandem with precision feeding strategies to maximize the efficiency and effectiveness of nutrient utilization by cattle, specifically for improving growth, feed efficiency, and production sustainability (Tedeschi et al., 2003). In conclusion, we believe that the factors limiting the practical use of precision feeding can ultimately be alleviated through the accumulation of knowledge and technology development from continued research, although the logistical hurdles of implementation at the feedyard-level remain to be overcome. Addressing these limitations and overcoming these hurdles to the adoption of precision feeding will help to maintain or improve animal performance while minimizing NH_3 emissions.

6.1.2 Manipulation of crude protein concentration and protein type

NH_3 emissions from beef cattle feedyards are sensitive to dietary CP concentrations (Cole et al., 2005; Todd et al., 2006, 2011). Approximately 25 to 50% of N intake is lost into the atmosphere as NH_3 when beef cattle are fed CP to meet their physiological and growth needs (Hristov et al., 2011; Todd et al., 2013). By reducing dietary CP content to match animal needs more closely, more urea recycling is stimulated, overall feed efficiency improves, and N losses are minimized (Galles, 2011). In previous studies, it was reported that NH_3 emission, as a consequence of volatilization of excreted N, could be reduced by a maximum of 67% when CP% in the feed was adjusted to around 10-11% from 13-16% (Table 6) in some situations. According to Menezes et al. (2016), the CP% of the diet for Nellore bulls was reduced from 14% to 10%, resulting in no significant difference in animal performance and carcass characteristics. However, it is important to note that reducing dietary CP levels below that required to meet the N needs of rumen fermentation and the AA needs of the animal would be expected to reduce growth performance and feed efficiency under most scenarios (Cole et al., 2005; Proctor, 2023). As a similar strategy to avoid these concerns, the method of phase feeding and oscillating dietary CP, which is the approach to more accurately apply CP according to the growth stage of livestock or environment, has been reported to mitigate N excretion and NH_3 emissions (Table 6).

In addition, manipulating the type of protein source in the diet can be helpful to mitigate N losses in manure. There are two types of protein: rumen degradable protein (RDP) and rumen undegradable protein (RUP). RDP is the protein broken down by the microbes in the rumen and used for microbial growth. RUP is the protein that escapes fermentation in the rumen and is digested in the small intestine. In beef cattle, 40 to 80% of non-retained N is excreted in the urine, and this quantity typically increases as dietary CP and RDP concentrations increase in the diet (NASEM, 2016). Therefore, N excretion can be reduced by increasing the proportion of RUP from the protein source required to satisfy the protein requirements (RDP+RUP) of cattle in the diet. However, it is important to ensure that RDP levels are high enough to satisfy the N requirement of the rumen microorganisms, as a deficiency would be expected to decrease the extent of fermentation and ultimately increase NH_3 emission intensity due to decreased feed efficiency. Increasing RUP level is also expected to increase N utilization efficiency by enhancing urea recycling to compensate for rumen microbial requirements due to RDP deficiency. Cole et al. (2005) reported that as RUP among the protein sources in the diet increased, the N excretion emitted from the urine decreased, and it affected the actual mitigation of NH_3 emissions. In addition, Batista et al. (2016) reported that corresponding with increased N intake, urinary N excretion was greater with supplementation, but supplementation did not affect fecal N excretion. Nevertheless, in response to RUP, fecal N excretion linearly increased, but urinary N excretion was not affected. In terms of NH_3 mitigation in ruminants, it is considered the reduced urinary N excretion will have a more positive effect on mitigating NH_3 emissions than the increased fecal N excretion by replacing RDP with RUP. However, there are a few things to consider with RUP. Batista et al. (2016), in their meta-analysis, reported that for diets with around 15% CP, a decrease in the efficiency of the incorporation of recycled N into ruminal microbial N was observed, with an efficiency of around 21%. This indicates that the efficiency of recycled N use is lower from RUP than the efficiency of consumed N use from RDP. Feeding RUP above requirements directly causes N excretion in manure. Also, to take advantage of efficient N recycling as RUP increases, the diet must be formulated to meet the energy requirements. This suggests that without an appropriate energy supply, enhanced N reuse from rumen microorganisms may not be obtained due to the lack of the carbon resources required for microbial protein production, leading only to increased N excretion rather than production of protein sources. In conclusion, decreasing CP concentration in the diet can potentially decrease NH_3 emissions, although it also decreases average daily gain, which can increase days on feed, the amount of manure deposited in the pens, and consequent increase in NH_3 emissions. Therefore, careful diet manipulation is needed to avoid unintended negative consequences for animal production and the environment.

6.1.3 Growth-promoting technologies

Growth-promoting technologies (implants and feed additives) are commonly used to reduce NH_3 emissions by less N excretion through increasing the efficiency of energy use for growth and by low cumulative NH_3 emissions from fewer days on feed required to reach finished weight. Although the specific mechanism for increasing productivity by growth-promoting technologies in beef cattle is different, growth-promoting technologies such as hormone implants and ß-adrenergic agonists increase nutrient use for protein synthesis and indirectly lead to decreased lipogenesis (Hutcheson et al., 1997; Nichols et al., 2002; Lean et al., 2014). It was reported

TABLE 6 Evaluation of management practices to mitigate NH_3 emission in the pre-excretion stage.

Reference	Management type	Measurement or Estimation Method		Application dose	NH_3 or N Excretion		*p* value
		NH_3 concentration	NH_3 emission	Range Value (% DM)	Result	Rate (%)	
Erickson et al. (2000)	CP manipulation	N mass balance		13.4 to phase-fed (10.5–12.0)	158 to 108 g/head/d	32	0.01
Pandrangi et al. (2003)	CP manipulation	Acid trap	Flux chamber	13.0 to 11.0	1.69 to 0.79 g/ m^2/d	53	<0.05
Cole et al. (2005)	CP manipulation	Acid trap	Flux chamber	13.0 to 11.5	1.95 to 1.24 g/ m^2/d	37	<0.01
Todd et al. (2006)	CP manipulation	Acid trap	Flux chamber	13.0 to 11.5	0.18 to 0.10 g/ m^2/d	44	<0.01
	CP manipulation	Acid trap	Integrated Horizontal flux		0.29 to 0.22 g/m^2/ d (Spring data)	24	<0.01
Cole et al. (2006)	CP manipulation	N mass balance		13.0 to 10	5.2 to 1.7 g/head/d	67	<0.01
Archibeque et al. (2007)	CP manipulation	N mass balance		Feeding CP 13.9% vs Oscillating feeding of low (9.1%) and high (13.9%) at 48h intervals	59.6 to 39.7 g/d	33	<0.01
Quinn et al. (2007)	CP manipulation	N mass balance		14.2 to phase-fed (avg 12.1%)	150 to 109	27	0.02
	CP manipulation	N mass balance		12.3 to phase-fed (avg 12.5%)	92 to 76	17	0.11
Erickson and Klopfenstein (2010)	CP manipulation	Meta-Analysis		13.6 to phase-fed (avg 11.5%)	158 to 108	32	0.01
	CP manipulation	Meta-Analysis		13.4 to phase-fed (avg 11.7%)	73 to 62 g/ head/day	15	0.32
Galles et al. (2011)	CP manipulation	Acid trap	Flux chamber	13.5 to 11.6 (for 45 days)	7.1 to 3.7 g/m^2/d	48	<0.10
Todd et al. (2013)	CP manipulation	Open path laser	Meta-Analysis Inverse dispersion model	16.0, 13.5, and 11	169.9, 104.4 to 90.1 g/ head/day	47	N/A
Menezes et al. (2016)	CP manipulation	N mass balance		14.0 to 10.0% DM	130.3 to 93.2 g/d	28	<0.01
Mejia Turcios (2024)	CP manipulation	Cattle pen enclosures		-150 g/head/d on rumen available protein to microbial crude protein ratios (RAP: MCP) vs. +150 g/head/d on RAP: MCP.	N/A	52	<0.01
Stackhouse et al. (2012)	Growth promoting technologies	Optical sensors (Innova 1412 and TEI 55C)	Flux chamber	33.1 mg/kg DM of monensin, 12.2 mg/kg DM of tylosin phosphate 8.3 mg/kg of DM of zilpaterol hydrochloride implantation with a combination of 120 mg trenbolone acetate and 24 mg estradiol	109 to 63 g/head/day	42	<0.01
Ross (2021)	Growth promoting technologies	N/A		Finishing ration containing 27.3 g ractopamine/907 kg dry matter	N/A	17	0.03
Aboagye et al. (2022)	Growth promoting technologies	N mass balance		Implanted 120 mg of trenbolone acetate, 24 mg of estradiol USP, and 29 mg of tylosin tartrate	51 to 46 g/ head/day	10	N/A

(Continued)

TABLE 6 Continued

Reference	Management type	Measurement or Estimation Method		Application dose	NH₃ or N Excretion		p value
		NH_3 concentration	NH_3 emission	Range Value (% DM)	Result	Rate (%)	
Wendler et al. (2025)	Growth promoting technologies	N mass balance		Optaflexx (ractopamine hydrochloride, 300 mg/head/day for 35 d) and Experior (lubabegron fumarate, 36 mg/head/day for 56 d + 4 d removal)	3338 to 3126 g cumulative NH_3	5-14	<0.01

N/A, not available; 6.2 Ammonia mitigation practices in the post-excretion stage.

that implants enhance both ADG and feed conversion, while implanted cattle often have less marbling and lower quality grades (Preston and Herschler, 1992; Selk, 1999; Ohnoutka et al., 2021). Also, monensin, which is generally included as a growth-promoting technology, is an ionophore antimicrobial that increases overall energy yield from feed and improves animal growth performance by increasing the ratio of propionate to acetate and decreasing the deamination of amino acids through preferentially inhibiting gram-positive bacteria in the rumen (Perry et al., 1976; Russell and Strobel, 1988; Tedeschi et al., 2003). Also, it prevents and controls Coccidiosis caused by Eimeria ssp in ruminants. An increase in protein synthesis with growth-promoting technologies would be expected to reduce N excretion. It has been reported that the use of conventional productivity-enhancing technologies (combination of implant, monensin, tylosin, ß-adrenergic agonists, and others), mitigated NH_3 emissions by 10~42%, but the effect of only implants mitigated 17% of NH_3 emissions (Stackhouse et al., 2012; Ross, 2021; Aboagye et al., 2022; Table 6). Additionally, the effect of only ß-adrenergic agonists reduced NH_3 emissions by 5 to 14% (Wendler et al., 2025; Table 6). The detailed mechanisms and effects of each growth-promoting technology are summarized in Brandani et al. (2023).

6.2 Ammonia mitigation practices in the post-excretion stage

6.2.1 Manure amendments

Manure amendment can be divided into chemical and physical amendments (Brandani et al., 2023). The NH_3 mitigation mechanism of chemical amendments is to add chemical compounds to manure to suppress the hydrolysis of excreted urinary urea or to create an environment with low pH, which is an unfavorable condition for NH_3 volatilization to occur. Representative examples of chemical amendments include urease inhibitors, N-(n-Butyl) thiophosphoric triamide, calcium chloride, humate, and aluminum sulfate. Physical amendments, such as biochar, carbon-rich material or biomass, bentonite, viscous plastic clay, and zeolite, microporous, crystalline aluminosilicate materials, act to adsorb NH_3 before being released into the atmosphere from the pen surface (Brandani et al., 2023).

Evaluation of manure amendment on the open feedyard surface to mitigate NH_3 emission has shown a wide range (19 to 98%) in mitigation effectiveness (Table 7). The urease inhibitor reduced NH_3 emissions by 26–66% on the manure surface in lab and pilot-scale studies but did not show significant mitigation at the field scale. However, nitrogen fertilizers coated with the urease inhibitors showed a significant mitigation of NH_3 emissions on grassland (67-79%). This suggests that further research is needed to determine the best application methods for urease inhibitors to achieve significant NH_3 reduction in feedyard manure. The effects of calcium chloride, humate, and aluminum sulfate, which lower the pH and inhibit urease decomposition, resulted in a mitigation rate of 20 to 71% (Shi et al., 2001; Spiehs and Woodbury, 2022) at the lab and pilot scale. As physical amendments, the lignite showed a mitigation rate of 66% (Chen et al., 2015) at the pilot scale. In addition, the mixture of biochar and bentonite showed a mitigation rate of 43%, and a 3% addition of zeolite reduced 10% of NH_3 emission (Szymula et al., 2021) at the lab scale. However, the low cost-efficiencies and the negligible financial benefit of NH_3 suppression made it challenging for manure amendment to be widely adopted in the feedyard. There is a growing need for technology that is economically advantageous and can be easily adapted to mitigate NH_3 emissions in the beef cattle feedyard effectively and at scale. In this respect, water application using sprinklers is one option exhibiting some promise (Lupis et al., 2012).

6.2.2 Water application

Water sprinklers are recognized to decrease dust emissions and have been adopted by some to mitigate heat stress for cattle, but they have not been used to mitigate NH_3 from the beef cattle feedyard (Brandani et al., 2023). The mechanism of water application to mitigate NH_3 comes from a dilution effect, which could relate to the simple leaching of aqueous NH_4^+ away from the surface or absorption of volatilized NH_3. Hutchinson et al. (1982) suggested that precipitation events cause a dramatic increase in the size of the reservoir available for NH_3 to exist in solution-diluting the NH_4^+ concentration and effectively decreasing the area of the air/water interface in the manure, the boundary at which volatilization occurs (40% NH_3 mitigation). However, a significant increase in NH_3 (160% NH_3 generation) was reported two days after precipitation, which leads us to question the temporal impact and whether water application is actually effective in mitigating NH_3 emissions in time scales relevant to feedyard management. Therefore, there is still a concern that it may cause more NH_3 volatilization in the long term by increasing the microbially mediated production of aqueous NH_3 within the water-filled pore space of the manure on the pen surface (Lee

TABLE 7 Evaluation of management practices to mitigate NH_3 emission in the post-excretion stage.

Reference	Management type	Measurement or Estimation Method		Application dose	NH_3 or N Excretion		p value
		NH_3 concentration	NH_3 emission	Range Value (% DM)	Result	Rate (%)	
Varel et al. (1999)	Urease inhibitor (N-(n-butyl) thiophosphoric triamide; NBPT)	Acid trap	N mass balance (Kjeldahl digestion)	22.8 kg/ha once per week for 42 days	5.0 to 2.1 g/kg manure (by 35 days)	58	N/A
Shi et al. (2001)	Urease inhibitor (NBPT)	Acid trap	Flux chamber	1 kg/ha for 21 days	4 to 1.44 g NH_3-N	65	<0.05
	Urease inhibitor (NBPT)	Acid trap	Flux chamber	2 kg/ha for 21 days	4 to 1.37 g NH_3-N	66	<0.05
	Surface amendment (Calcium chloride)	Acid trap	Flux chamber	9000 kg/ha	4 to 0.9 g NH_3-N	78	<0.05
	Surface amendment (Humate)	Acid trap	Flux chamber	9000kg/ha	4 to 0.9 g NH_3-N	68	<0.05
	Surface amendment (Aluminum sulfate)	Acid trap	Flux chamber	9000kg/ha	4 to 0.7 g NH_3-N	98	<0.05
	Surface amendment (commercial product, Ammonia Hold, Lonoke, Arkansas)	Acid trap	Flux chamber	750 kg/ha	4 to 2.7 g NH_3-N	32	<0.05
Chen et al. (2015)	Surface amendment (Lignite)	Chemiluminescence	Integrated Horizontal flux	4.5 kg/m^2	156 to 53 g NH_3-N/head/day	66	N/A
Szymula et al. (2021)	Surface amendment (Biochar)	Berthelot reaction method		3% addition of biochar	18 to 11 mg/L	41	<0.05
	Surface amendment (Zeolite)	Berthelot reaction method		3% addition of zeolite	18 to 17 mg/L	9	N/S
	Surface amendment (Mixture of bentonite and zeolite)	Berthelot reaction method		3% addition of a mixture of bentonite and zeolite	18 to 13 mg/L	28	<0.05
Spiehs and Woodbury (2022)	Surface amendment (Aluminum sulfate)	Acid trap	Flux chamber	300g/6 kg of manure + water Data from 0 to 7 days	Approximate 43 to 33 mg/m^2/h	~20	<0.05
				600g/6 kg of manure + water Data from 7 to 14 days	Approximate 26 to 10 mg/m^2/h	~60	<0.05
Parker et al. (2004)	Urease inhibitor (NBPT)	Acid trap	Flux chamber	1 kg/ha	26 to 13 μg/m^2/s	49	<0.05
	Urease inhibitor (NBPT)	Acid trap	Flux chamber	2 kg/ha	26 to 9 μg/m^2/s	68	<0.05
Parker et al. (2011)	Urease inhibitor (NBPT)	Acid trap	Flux chamber	5 kg/ha initially and then doubled every 4 days to a maximum of 40 kg/ha	40 to 12 μg/m^2/s	73	<0.05
	Urease inhibitor (NBPT)	Acid trap	Flux chamber	5 kg/ha	40 to 11 μg/m^2/s	70	<0.05
Dawar et al. (2011)	Urease inhibitor (NBPT)	Acid trap	Flux chamber	Urea coated with NBPT at 0.1% (w/w) of urea	19 to 6 kg/ha	69	<0.05
Parker et al. (2016)	Urease inhibitor (NBPT)	Chemiluminescence	Flux chamber	1, 2, 4, 8, and 40 kg/ha	31 to 30 μg/m^2/s (40 kg/ha data)	4	N/S

(Continued)

TABLE 7 Continued

Reference	Management type	Measurement or Estimation Method		Application dose	NH$_3$ or N Excretion		p value
		NH$_3$ concentration	NH$_3$ emission	Range Value (% DM)	Result	Rate (%)	
Forrestal et al. (2016)	Urease inhibitor (NBPT)	Acid trap	Wind tunnels	40 kg N/ha of urea + NBPT	N/A	79	<0.05
Krol et al. (2020)	Urease inhibitor (NBPT)	Acid trap	Integrated Horizontal flux	20, 30, 40 kg N/ha of urea + NBPT and urea+ NBPT + NPPT	N/A	67	N/A
Hutchinson et al. (1982)	Water application	Acid trap	Vertical gradient flux model	60 mm precipitation	42 to 25 μg/m^2/s	40	N/A
				After precipitation, surface drying for 2day	25 to 65 μg/m^2/s	Increased 160	N/A
Todd et al. (2005)	Water application	Acid trap	Flux gradient model	Precipitation (Dose: N/A)	93 to 55 μg/m^2/s	41	N/A
Pandrangi et al. (2003)	Water application	Acid trap	Flux chamber	270 mL of water (at 9 day)	8~16 to 10~18 μg/m^2/s	Increased 26	N/A
Saarijärvi et al. (2006)	Water application	Passive-diffusional samplers	Flux chamber	20 mm of water	25 to 11 μg/m^2/s	56	N/A
Galles (2011)	Water application	Acid trap	Flux chamber	5 mm of water	451 to 335 μg/m^2/s (for 1 day data)	27	<0.01
Parker et al. (2011)	Water application	Acid trap	Flux chamber	173 mL of water	40 to 25 μg/m^2/s	37	<0.05
Lee et al. (2023)	Water application	EC sensor	Flux chamber	5 mm of deionized water	36 to 39 μg/m^2/s (by 4 days data)	Increased 8	<0.01

N/A, not available; N/S, not significant.

et al., 2023). As proof of this, a few studies have reported results contrary to the NH$_3$ mitigation with the water application (Table 7).

In summary, there is scientific agreement that the water application may mitigate NH$_3$ emissions (27~56%) under carefully controlled conditions and over short time scales. However, because of the lack of consensus on the use of water application, there is a concern that the NH$_3$ mitigation due to water sprinkling is temporary and generates more NH$_3$ during the evaporation process, especially when rapid evaporation of water occurs due to hot, windy weather. The impact of the water application on NH$_3$ emissions continues to be investigated and a clearer interpretation of this is expected to emerge in the future.

7 Discussion

The current major hurdle facing cattle feedyards in applying the above BMPs solely for NH$_3$ mitigation is whether the practically achievable benefits justify their costs. To be specific, feed composition is made close to the requirements of cattle with safety margins, and the pre-excretion technologies (e.g., growth-promoting technologies) are used to increase the nutrient-use efficiency of cattle, minimizing the nutrient excretion in most feedyards. According to Legesse et al. (2018), through such improvements in livestock management and in reproductive efficiency, NH$_3$ (kg) emitted per beef (kg) decreased 20% from 1981 to 2011. However, some studies have reported that the expansion of large-scale intensive livestock operations, such as CAFOs, has contributed to increasing total NH$_3$ emissions (Legesse et al., 2018; Schultz et al., 2019; Wyer et al., 2022). Therefore, to mitigate NH$_3$ emissions, higher-precision feeding and active use of pre- and post-excretion practices are necessary. However, overly strict implementation of precision feeding strategies may introduce unintended variability in livestock performance and increase operational costs due to reduced safety margins and the need to modify existing feedyard infrastructure. In addition, post-excretion BMPs constitute essentially unrecoverable expenses unless the BMP facilitates the production of a marketable product. Therefore, the benefits of the practices implemented to mitigate NH$_3$ emissions while bearing additional costs are an important factor in the feedyard's decision to implement BMP.

High ambient NH$_3$ concentrations (average 42 ppm) have been reported to have a negative impact on the bovine lungs in respiration chamber-scale experiments, leading to increased total white cell and mononucleated cell counts ($p < 0.05$, Accioly et al., 2004). However, a knowledge gap exists on the effect of ambient NH_3 concentrations in the feedyard on animal productivity. Reported background concentrations of NH_3 at feedyards typically range from<1 to 2000 μg/m^3 (0–3 ppm at 25 °C, 1 atm; Todd et al., 2005; Hristov et al., 2011). However, as highlighted by Hristov et al. (2011), the concentration of atmospheric NH_3 is highly variable in various forms (gas, particulate, and liquid) and depends on the presence of other compounds. Consequently, NH_3 concentrations are subject to considerable variability due to a combination of environmental and feedyard management factors, and under certain conditions, concentrations may reach levels that can potentially affect animal productivity. Additional research is needed to evaluate potential productivity improvement and its link to mitigating NH_3. Such studies could support the necessity of NH_3 mitigation efforts and offer practical benefits for livestock operations.

Based on the results currently reported, the following additional benefits can be considered for the use of BMP related to NH_3 mitigation. Precision feeding and diet manipulation aims to provide nutrient supply more precisely with the nutrient requirements, thus the benefits include economic returns through reduced excretion to the environment and improved efficiency of resource utilization by leading to decreased feed intake and thereby decreased enteric CH_4 emissions (Zuidhof, 2020; Galyean and Hales, 2023). Growth-promoting technologies increase the efficiency of energy and nutrient use, thereby increasing animal productivity and mitigating environmental effects while reducing the amount of time required to finish cattle. As an example, ionophores, one of the growth-promoting technologies, may decrease protein degradation in the rumen, increase feed protein utilization, and reduce N losses (Tedeschi et al., 2003). In addition, it can decrease feed intake (4%) without affecting animal performance, and mitigate 25% of enteric CH_4 emissions (Tedeschi et al., 2003).

In the case of manure amendment, it is not directly related to animal performance, but it is related to benefits for manure value (C:N ratio) and the mitigation of other gases (H_2S, GHGs, and VOCs). The C:N ratio in manure could vary greatly depending on diet, manure storage, manure management, and feedyard environments. It is generally reported that the C:N ratio of beef and dairy manure is 10 to 15:1 (Okopi et al., 2024). While close to the optimal C:N ratio (20 to 30:1) for net N mobilization through soil microorganisms (Hadas et al., 1992), manure C is insufficient in most cases. Manure amendments, which are a C source and particularly physical amendment, can improve C:N ratio and a MC (50-70%) for composting and land application. In addition, manure amendments have been reported to be effective in mitigating various gas emissions from cattle manure (Wheeler et al., 2011; Spiehs et al., 2019; Kaikiti et al., 2021; Chen et al., 2024). However, manure amendment could cause secondary air pollution from physical amendment (e.g., PM from biochar, Gelardi et al., 2019) or chemicals (e.g., CH_4 and H_2S from aluminum sulfate, Spiehs et al., 2019) added to prevent NH_3 gas volatilization. Since research on manure amendments has focused on target gas mitigation, there has been little research on the generation of by-products or gases after application (Maurer et al., 2016). It is important to be cautious when using the amendments to avoid secondary, perverse effects.

Lastly, the practice of water application was proposed as a method to reduce heat stress in terms of animal production, but it could potentially improve feed efficiency during the summer (Mader and Davis, 2004). Water application may be a cost-effective solution for industry PM control in some circumstances (Yonkofski et al., 2019), and it has been reported to have the mitigation effect of other gases (GHGs such as CH_4 and N_2O) as well (Parker et al., 2021). Precipitation, which is the natural way to apply water, was observed to mitigate the emission of CH_4 and N_2O below detection levels for several days after the precipitation event in the feedyard (Parker et al., 2021). In lab-scale experiments, increased N_2O emission has been observed after precipitation for several days (Parker et al., 2017, 2018), but this phenomenon has not been observed on the field scale (Parker et al., 2021). Further research is still needed because there are concerns about more gas volatilization during the drying process after water application and practical research is necessary into how water can be applied to feedyards as precipitation to achieve beneficial effects.

The direction we should take to mitigate NH_3 emissions in feedyards is to maximize N-use efficiency of beef cattle by optimizing the pre-excretion management while simultaneously minimizing the environmental impacts using post-excretion management. To encourage the adoption of a given management practice, more research is needed to quantify its benefits, to describe as fully as possible the conditions under which those benefits may be realized in practice and at scale, to develop new promising practices, and to reckon transparently with a practice's perverse effects, if any.

8 Conclusion

NH_3 emitted from beef cattle feedyards is a high-profile environmental concern because of health hazards, its contribution to fine particulate formation, and contamination of air and surface waters. Mitigation of NH_3 emissions addresses social concerns, minimizes the risk of undesirable environmental events, and is important to the sustainability of the beef industry. In this review, we reported the state of the science concerning NH_3 emissions from beef cattle feedyards, methods for quantifying NH_3 emissions, NH_3 EF, and some management practices to mitigate NH_3. Ammonia emissions primarily come from urinary urea in cattle manure on feedyard surfaces. A significant portion of the N in the manure is converted to NH_4^+ and is eventually volatilized to the atmosphere as NH_3. In the past, constant EFs were used to inventory NH_3 emissions. Currently, NH_3 EF estimated by process-based mechanistic models reflecting various factors affecting NH_3 emissions in the feedyard environment are available. As process-based mechanistic models, the backward Lagrangian stochastic model was widely used to convert NH_3 concentration measurements into emissions in the beef cattle feedyard. This review of current literature indicated the average NH_3 emissions from the cattle feedyard as 119 g/head/day (ranging from 24 to 318 g/head/day), and the average NH_3 flux rate as 58 μg/m^2/s (ranging from 2 to 185 μg/m^2/s). Although it is agreed that more realistic NH_3 EF are being obtained using process-based models, there is still significant variation of estimated NH_3 EF depending on the diet composition, the manure management, and the feedyard environment. We note the need to improve inventories of NH_3 emissions into categories of manure management implemented and feedyard environment. Some mitigation strategies can be effective, such as manipulating the diet to reduce N excretion, increasing animal performance with growth-promoting technologies, and using manure amendments. Of those, precision diet feeding to meet, but not exceed, protein requirements appears to be the most practical way to reduce N losses. However, careful diet manipulation and additional research are needed to avoid unintended negative consequences for animal production.

Author contributions

ML: Data curation, Formal Analysis, Investigation, Visualization, Writing – original draft, Writing – review & editing. BA: Conceptualization, Funding acquisition, Methodology, Project administration, Resources, Supervision, Validation, Writing – original draft, Writing – review & editing. LT: Methodology, Resources, Validation, Writing – review & editing. JK: Methodology, Resources, Validation, Writing – review & editing. CB: Resources, Validation, Writing – review & editing. VG: Resources, Validation, Writing – review & editing. JS: Resources, Validation, Writing – review & editing. KC: Resources, Validation, Writing – review & editing.

Funding

The author(s) declare that financial support was received for the research and/or publication of this article. This research was funded by the NRCS Conservation Innovation Grant (project number NR213A750013G037), and additional support was provided by the Colorado Livestock Association.

Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Supplementary material

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fanim.2025.1608387/full#supplementary-material

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