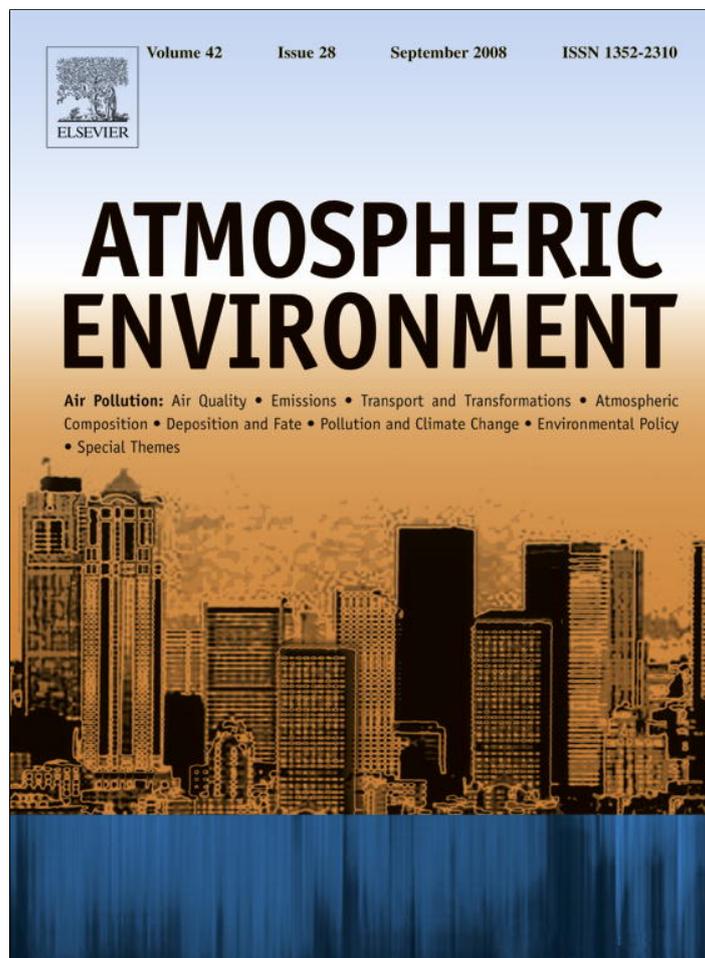


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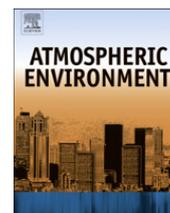
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Ammonia emissions from a beef cattle feedyard on the southern High Plains

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ABSTRACT

Concentrated animal feeding operations (CAFOs) are major sources of ammonia emitted into the atmosphere. There is considerable literature on ammonia emissions from poultry and swine CAFO, but few comprehensive studies have investigated large, open lot beef cattle feedyards. Ammonia emission rates and emission factors for a 77-ha, 45 000-head commercial beef cattle feedyard on the southern High Plains were quantified using measured profiles of ammonia concentration, wind speed and air temperature, and an inverse dispersion model. Mean summer emission rate was 7420 kg NH₃ d⁻¹, and winter emission rate was about half that, at 3330 kg NH₃ d⁻¹. Annual NH₃-N emission rate was 4430 kg NH₃-N d⁻¹, which was 53% of the N fed to cattle. Daily per capita NH₃-N losses increased by 10–64% after the daily per capita N in feed rations increased by 15–26%. Annual emission factors for the pen area of the feedyard were 19.3 kg NH₃ (head fed)⁻¹, or 70.2 kg NH₃ Mg⁻¹ biomass produced. Annual emission factors for the retention pond of the feedyard were estimated to be 0.9 kg NH₃ (head fed)⁻¹, or 3.2 kg NH₃ Mg⁻¹ biomass produced.

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1. Introduction

Human activity has more than double the amount of reactive nitrogen that cycles through terrestrial ecosystems (Smil, 1990; Vitousek et al., 1997), with many negative impacts on ecosystem function and health, and air quality. Estimates of the contribution of agriculture to reactive N in the environment range from 50 to >90%, with animal agriculture contributing the majority (Bouwman et al., 1997; Ferm, 1998; Galloway and Cowling, 2002; Howarth et al., 2002).

Ammonia volatilized to the atmosphere is a major path for fugitive reactive N. Ammonia is the predominant base in the atmosphere and readily reacts with acidic compounds

like sulfate or nitrate to form particulates with mean aerodynamic diameter of 2.5 μm (PM_{2.5}). This class of particulates is of concern because they are respirable and have been implicated in human respiratory problems. Ammonia is not a criteria pollutant under U.S. Environmental Protection Agency (U.S. EPA) regulations, but is of interest because it is a precursor to PM_{2.5} formation.

Concentrated animal feeding operations (CAFO) are major sources of ammonia emitted to the atmosphere. There is considerable literature on ammonia emissions from poultry and swine CAFO, but few comprehensive studies have investigated large, open lot beef cattle feedyards. Hutchinson et al. (1982) was one of the first studies to quantify ammonia emissions from a commercial feedyard. Researchers in Nebraska used a mass balance approach to quantify N at various points in the feedyard system, and calculated N volatilization losses as the

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residual of the N balance (Bierman et al., 1999; Erickson et al., 2000; Erikson and Klopfenstein, 2001). Micrometeorological methods such as the flux-gradient method (Baek et al., 2006; Todd et al., 2005) or an inverse dispersion model (Flesch et al., 2007) were employed to quantify ammonia emissions from a Texas cattle feedyard.

The southern High Plains cattle feeding industry feeds over 7 million head of cattle a year, about a third of the United States total. Most are fed in more than 100 open lot feedyards with capacities that range from 5000 to more than 100 000 head, with median capacity of 30 000 head (Eck and Stewart, 1995; SPS, 2000). Calves typically enter feedyards at 250–300 kg and are fed for 150–180 days to a final weight of about 550 kg. Cattle are fed corn-based (70–80%) diets, optimally with 13–13.5% crude protein (CP) and often supplemented with urea (Cole et al., 2005).

There are two main sources of ammonia on feedyard surfaces; NH_4^+ hydrolyzed from urea in urine, and NH_4^+ mineralized from more complex organic forms, predominantly in feces. Urea is relatively quickly hydrolyzed, commonly within hours of excretion, and provides a pool of NH_4^+ that is continuously replenished as cattle urinate (Varel, 1997; Petersen et al., 1998; Arogo et al., 2001). Nitrogen excreted in urine ranged from 30 to 80% of fed N, and typically increases as crude protein in a diet increases beyond the physiological needs of animals (Cole et al., 2005; Erickson et al., 2000; James et al., 1999; Smits et al., 1995; Todd et al., 2006). In contrast, mineralization is a much slower process and provides a more constant, slow rate source of NH_4^+ .

Micrometeorological methods used to determine gaseous emissions to the atmosphere are advantageous because they do not interfere with the processes of emissions and they integrate emissions over areas on the scale of entire feedyards (Fowler et al., 2001; Harper, 2005; McGinn and Janzen, 1998). Generally speaking, micrometeorological methods rely on measurements in and characterization of the atmosphere near the ground. Quantifying ammonia, or any other gaseous emissions, from beef cattle feedyards entails two major challenges: (i) measurement of atmospheric NH_3 ; and (ii) relating that concentration measurement to a surface emission rate based on the dispersive state of the atmosphere.

Atmospheric dispersion models describe the relationship between a source of a gas and a downwind concentration (Harper, 2005). These models require assumptions about the strength of the wind and turbulence, and are most commonly used to predict the gas concentration downwind of a known emission source (forward mode). However, in the context of our study, these models can be used to infer the emission rate if given the concentration at a downwind sensor (backward mode). Many different types of dispersion models could be used for this “inverse” analysis (e.g. Gaussian plume model, K-theory model), but Flesch et al. (2004) described how backward Lagrangian stochastic (BLS) models are well suited to problems where the source-to-sensor dispersion takes place within the atmospheric surface layer, i.e. short-ranges. The BLS model infers the flux rate from a defined source by modeling the upwind trajectories of an ensemble of tracer gas particles

from where concentration is measured back to the source area, using a Monin–Obukhov Similarity Theory parameterization of the wind field (Flesch and Wilson, 2005). There is flexibility in where and how tracer gas concentration is measured. Inputs to describe the wind conditions include friction velocity, surface roughness length, Monin–Obukhov length and wind direction; these can be determined from three-dimensional sonic anemometer measurements, wind speed and air temperature profiles with a wind vane, or a single wind speed measurement with qualitative atmospheric stability estimates. The BLS model has been tested and compared positively to other methods for estimating fluxes of methane (Laubach and Kelliher, 2005), ammonia (Sommer et al., 2005), and with gas release experiments (Flesch et al., 1995, 2004).

Our objectives were to (i) measure atmospheric ammonia, wind speed and temperature profiles at a typical commercial beef cattle feedyard for extended time periods in summer and winter; (ii) use these measurements as inputs into a BLS dispersion model to quantify ammonia emissions; and (iii) calculate ammonia emission factors for the feedyard based on annual ammonia emissions and cattle production.

2. Materials and methods

2.1. Site location and description and experimental campaigns

Research was conducted at a commercial beef cattle feedyard, established in 1967, located in the Texas Panhandle, with a total pen area of 77 ha (Fig. 1). Mean occupancy was 44 651 head, with an inverse stocking density of $17 \text{ m}^2 \text{ head}^{-1}$. Though the terrain is relatively flat, the feedyard surface is complex, with several small buildings, thousands of meters of 1.5-m tall pen fences, electrical poles, manure mounded in centers of pens, and

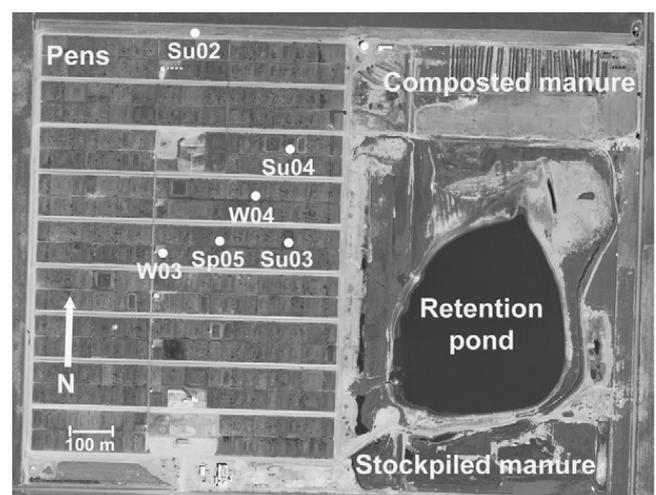


Fig. 1. Texas Panhandle commercial feedyard used in research. Pens (manure surfaces with cattle occupancy or recently occupied) covered 77 ha. Retention pond area was variable, depending on precipitation and runoff from pens; in this photo, pond area is 20 ha. Locations of meteorological towers during six campaigns are shown: Su = summer, W = winter, and Sp = spring. The season is followed by the year.

mobile cattle. A retention pond, manure stockpiles and compost rows were located east of the pens. Retention pond area was variable, depending on precipitation and runoff from pens, and ranged from 20 to 36 ha. Manure in pens was typically managed at the end of a 150–180 day feeding cycle by scraping manure from the pen perimeter and rebuilding manure mounds in the center of a pen and then removing excess manure. The semiarid climate of the region is characterized by hot summers and mild winters. Mean annual precipitation is 500 mm, with 75% falling from April through October. Potential evaporation is about 1500 mm, so that summer precipitation often rapidly evaporates. Prevailing winds are southerly to south-westerly, with wind direction almost half the time between 160° and 250°. The soil on which the feedyard was built is a Pullman clay loam (fine, mixed, superactive, thermic Tor-terric Paleustoll).

Six field campaigns were conducted, commencing in summer 2002 and ending in spring 2005 (Table 1). During each campaign, an instrument tower that held meteorological instruments and ammonia concentration measuring equipment was erected in a vacant pen (except summer 2002; see Fig. 1). The location of the tower changed from year to year, depending on expected seasonal prevailing wind directions, necessities for power, and feedyard management. Nitrogen fed to cattle during each campaign was calculated using total head count, total feed fed, and dry matter and nitrogen composition of the diets fed from data provided by the feedyard and from analysis of dietary samples collected during each campaign.

2.2. Tower measurements

Ammonia concentration was measured using acid gas washing samplers positioned on the tower at different heights for each campaign (Table 1), but sampling protocol was the same. Gaseous ammonia was trapped in gas washing bottles (Pyrex 250 ml, fritted cylinder with coarse porosity) by first drawing air through a Teflon filter to remove particulates, then bubbling it through an impinger in 80–120 ml of 0.1 N H₂SO₄. A greater volume of H₂SO₄ was used when greater evaporation was anticipated during warmer, drier conditions to ensure adequate volume of acid. Air flow rate of each gas washing bottle was measured with a precision, calibrated flow meter (Dry-Cal DC Lite, Bios International, Butler, NJ¹) at the beginning and end of each sampling period. Nominal air flow rate was 6 L min⁻¹. At the beginning of a sampling period, gas washing bottles with fresh acid were sealed and transported to the tower, exchanged with the bottles there, and sealed bottles with samples were returned to a mobile laboratory, where each sample was diluted to 100 ml with acid, 30 ml was decanted into a sample bottle, and then all samples were refrigerated until analysis. A calibrated flow injection analyzer (QuickChem FIA + 8000, Lachat Instruments, Milwaukee, WI) was used to quantify

ammonium concentration in the samples, with a minimum detection limit of approximately 10 µg L⁻¹. This corresponded to atmospheric ammonia concentrations of less than 1 µg m⁻³. However, experience indicated that the minimum detection limit of atmospheric ammonia was probably closer to 5 µg m⁻³. During the summer 2004 campaign, ammonia concentration was measured continuously using a chemiluminescence analyzer (17C, Thermo Environmental Instruments, Franklin, MA). Ammonia concentration at 3 and 6 m was measured sequentially using a three-way solenoid that switched gas sampling lines from one height to the other every 10 min. Data from the last 3 min out of each 10 min period were retained and averaged to allow for analyzer response time (Baek et al., 2006).

Profiles of wind speed and air temperature were measured at the same heights as atmospheric ammonia concentration. Cup anemometers (12102M, R.M. Young, Traverse City, MI) measured wind speed and aspirated, fine-wire (25.4 µm diameter) thermocouples (ASPTC, Campbell Scientific, Logan, UT) measured air temperature. Other meteorological measurements included incoming solar radiation (LI200X, Licor Inc., Lincoln, NE), relative humidity and air temperature (HMP45, Vaisala, Helsinki, Finland), wind direction (12005, R.M. Young, Traverse City, MI) and precipitation (TE525, Campbell Scientific, Logan, UT). Outputs from meteorological instruments were automatically recorded to a data logger (CR23X, Campbell Scientific, Logan, UT) that sampled instruments every 5 s and calculated 1-min means.

2.3. Inverse dispersion model

The backward Lagrangian stochastic (BLS) dispersion model used to estimate emissions was Windtrax (Thunder Beach Scientific, Nanaimo, Canada), a commercially available model. Details of the model theory, development and testing were given in Flesch et al. (1995, 2004, 2005) and Flesch and Wilson (2005). Ideally, inputs to the BLS model should have sample integration times of 15–120 min (Flesch et al., 1995, 2004) in order to meet the assumption of stationarity. Gas washing requires longer sample integration times. Sommer et al. (2005) pointed out this problem, especially with regard to atmospheric stability. They used passive ammonia samplers, with concentration integrated over time periods of 5–26 h and found that BLS flux estimates were within 16–24% of integrated horizontal flux estimates. They recommended the assignment of neutral stability for longer sample integration times that may violate the assumption of stationarity. Based on the calculated Monin–Obukhov lengths for this study (see below), the fraction of sampling times near neutral stability ($|L| > 100$) ranged from 55% during Trial 3 (summer) to 96% during Trial 2 (winter), when sampling time was 4 h during daytime and 16 h during nighttime. Over all the sampling periods in the study, 72% were near neutral stability.

For the first three campaigns, we adopted either 3-h (summers 2002 and 2003) or 4-h (winter 2003) daytime sample times. We found that in most cases, the calculated nighttime (9 h duration in summer and 16 h duration in winter) stability was near neutral. Daily mean emission for these three campaigns was the time-weighted mean

¹ The mention of trade names or commercial products in this article is solely for the purpose of providing specific information and does not imply recommendation or endorsement by the U.S. Department of Agriculture.

Table 1

Field campaign dates, sample integration times for ammonia concentration measurements, and profile heights for ammonia concentration, wind speed and air temperature

Campaign	Dates	Gas washing sample integration times (day/night) (h)	Profile heights (m)
Summer 2002	19 Aug–23 Aug	3/9	0.75, 1.5, 3, 6
Winter 2003	15 Jan–24 Jan	4/16	2, 3, 4, 5, 6
Summer 2003	14 Jul–1 Aug	3/9	2, 4, 6, 8
Winter 2004	26 Jan–6 Feb	2/2 ^a	1, 2, 4, 6, 8, 10
Summer 2004	14 Jun–6 Jul	1/1	3, 6
Spring 2005	28 Mar–12 Apr	2.5/2.5 ^b	3, 4, 5, 6

^a Sample time was 2 h out of every 4 h for 12 h during daytime; 2 h from 0000 to 0200 h during nighttime.

^b Samples were collected every 2.5 h for 10 h during daytime; 2.5 h from 0000 to 0230 h during nighttime.

of the sampling periods for a day. For the summer 2004 campaign, continuous sampling by the chemiluminescence analyzer allowed us to calculate hourly means. During the winter 2004 campaign, we adopted a 2-h on, 2-h off sampling schedule for 10 h during daytime, and a single 2-h nighttime sample that began at midnight. During spring 2005, sample times were 2.5 h for 10 h during daytime, with a single 2.5 h nighttime sample that began at midnight. Nighttime sampling for these two campaigns was triggered by a timer that turned on and off the vacuum pump of the gas washing system. The daily mean emission was integrated using the trapezoidal rule.

The feedyard pens source area was mapped into Windtrax as polygons defined by GPS coordinates. Pens were defined as manure surfaces either with cattle occupancy or recently occupied. Source areas of the pens for the six trials ranged from 76.9 to 77.33 ha (Table 2). Differences were because the vacant pen in which the tower was located was excluded as a source and the area of the tower pens varied. Work areas, service roads and feed truck alleys between pens were excluded from the source map. A sensitivity analysis performed with Windtrax on the effect of the retention pond indicated that it exerted negligible effect on measurements taken within the pens (data not shown). Therefore, the pond was excluded as a source area.

A BLS simulation for each sampling period was run at each measurement height, using measurements of wind speed and ammonia concentration from that height. Ammonia concentration and wind speed profiles were initially screened by plotting and checking for adherence to logarithmic profiles. An additional check on the validity of the concentration profiles and for adherence to the assumption of horizontal homogeneity was to run model simulations for each measurement height, and then to check the resulting flux estimates to see if flux remained

constant with height. A flux mean and standard deviation were calculated and coefficients of variation were carefully inspected. Other inputs were wind direction and Monin–Obukhov length (L). The Monin–Obukhov length is a stability length scale that expresses the contributions of mechanical and thermal turbulences (Prueger and Kustas, 2005). A single L was calculated from measurements of wind speed and air temperature profiles: first, the gradient Richardson number (Thom, 1975) was calculated; then, semi-empirical expressions relating L to Richardson number were applied (Högstrom, 1988, 1996). An ensemble of 50 000 particles was used for each simulation. A single roughness length (z_0) was used for all simulations; it was determined from a concurrent study during winter 2004 from 963 15-min sonic anemometer observations, with $z_0 = 0.09 \pm 0.12$ m, using the formulation given in Flesch et al. (2005). Input data were excluded when wind speed (used as a proxy for friction velocity) was less than 1.5 m s^{-1} and $|L| < 7$, in order to screen out conditions when Monin–Obukhov Similarity Theory was most likely to be violated (Flesch and Wilson, 2005). Flux rate was calculated by the model at each measurement height and then averaged to give the flux rate for a sampling period. Daily emission rate was calculated, as explained above, by either calculating the time-weighted mean of the emission rates for a day's sampling periods, or integrated using the trapezoidal rule (Kreyszig, 1972).

3. Results and discussion

3.1. Ammonia emission rates

There was no a priori reason to choose one height over another, so we opted to use all available wind, temperature and ammonia concentration data and to run simulations

Table 2

Feedyard cattle population, per capita nitrogen fed to cattle, per capita ammonia–N loss, and percentage of fed nitrogen lost as ammonia–N

Campaign	No. of days	No. of heads	Pen source area (ha)	Per capita fed N ($\text{g head}^{-1} \text{d}^{-1}$)	Per capita NH_3 -N loss ($\text{g head}^{-1} \text{d}^{-1}$)	NH_3 -N loss as % of fed N (%)
Summer 2002	5	42 804	77.33	162	117	72
Winter 2003	7	43 157	77.20	160	51	32
Summer 2003	10	48 463	77.27	186	130	70
Winter 2004	5	41 863	77.11	201	84	42
Summer 2004	12	49 109	76.90	205	131	64
Spring 2005	4	40 295	77.27	193	118	62

for each height. Any variability in model flux estimates would integrate measurement and modeling errors. We calculated a coefficient of variation (CV) for each set of profile flux estimates from a sampling time/model run and looked at their frequency distribution (Fig. 2). Over five trials (Trial 5 could not be used because there were only two heights), minimum CV ranged from 2.2 to 6.5%, mean CV ranged from 8.8 to 17.6% and maximum CV ranged from 26.5 to 60.2%. We concluded that uncertainty in the flux estimate probably was in the range of 9–18%. Of the 185 flux means, 77% of them had CV < 15%, indicating that most measurements were made in the fully adjusted layer and that flow was horizontally homogeneous.

During summer campaigns (June, July, August), with a total of 27 days, daily mean ammonia emission rate ranged from 5130 to 11090 kg d⁻¹ (Fig. 2). The overall mean ammonia emission rate (± standard deviation) during summer was 7420 ± 1580 kg d⁻¹. In 2004, DOY 167 and DOY 168 showed similar wind speed, relative humidity, and air and surface temperatures; however, late morning and midday ammonia concentrations were much greater on DOY 167 compared with DOY 168 (mean 860 µg m⁻³ versus 370 µg m⁻³, respectively), which accounted for the greater emission on DOY 167. Ammonia emission during winter (January, February, 12 days) ranged from 1910 to 4680 kg d⁻¹ (Fig. 3). The overall mean was 2670 kg d⁻¹ in 2003 and 4250 kg d⁻¹ in 2004, and averaged 3330 ± 1020 kg d⁻¹. Winter ammonia emission averaged 45% of the mean summer emission. A spring campaign in 2005 (late March and early April) yielded four complete days of data. Ammonia emission ranged from 3820 to 9280 kg d⁻¹, and averaged 5800 ± 2450 kg d⁻¹ (Fig. 3).

Daily per capita NH₃-N losses in summer, based on one-time capacity, were 117, 130, and 131 g head⁻¹ d⁻¹ in 2002, 2003 and 2004, respectively, and averaged (n = 27) 128 ± 25 g head⁻¹ d⁻¹ (Table 2; Fig. 4). During winter, one-time capacity per capita NH₃-N loss decreased, compared with summer losses, to 51 g head d⁻¹ in 2003 and 84 g head d⁻¹ in 2004; mean winter per capita loss was 64 ± 21 g head⁻¹ d⁻¹ (n = 12). One-time capacity per capita NH₃-N loss in spring 2005 averaged 118 ± 50 g head d⁻¹ (n = 4).

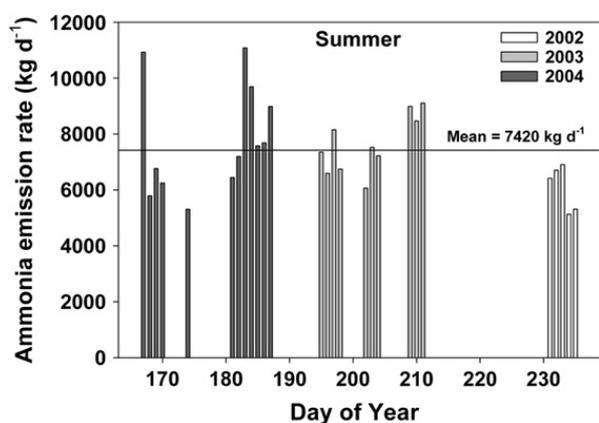


Fig. 2. Daily mean ammonia emission rate during summer for three campaigns conducted in 2002, 2003 and 2004.

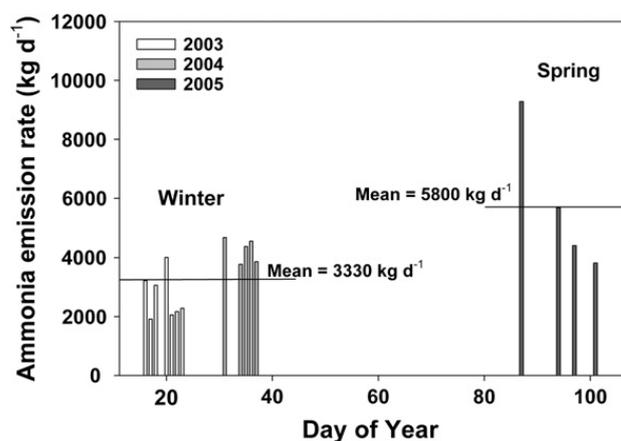


Table 3

Ammonia–N loss as a percentage of fed nitrogen from High Plains beef cattle feedyards

Study	Summer (%)	Winter (%)	Annual (%)
This study ^a	68	36	53
Flesch et al. (2007) ^a	63		
Harper et al. (2004) ^a	53	29	
Todd et al. (2005) ^{a,d}	45	44	45
Cole et al. (2006) ^{c,d}	51–65		
Erickson and Klopfenstein (2001) ^b	51–61		
Erickson et al. (2000) ^b	63		
Bierman et al. (1999) ^b	53–63	35	

^a Texas.

^b Nebraska.

^c New Mexico.

^d Gaseous N loss, based on change in N:P ratio of feed and manure.

(2006) found that gaseous N loss ranged from 51 to 65% of fed N, over a range of dietary CP in a comprehensive New Mexico cattle feeding trial. Annualized ammonia–N loss found in this study was 53%.

From 30 to 70% of N in cattle diets is routinely excreted as urinary N (Cole et al., 2006). This value increases when fed N exceeds animal requirements, and can contribute to ammonia volatilization. For example, Todd et al. (2006) found that increasing dietary CP from 11.5 to 13% increased ammonia emission from an artificial feedyard surface by 39%. Cole et al. (2006) reported that when CP increased from 11.5 to 13%, apparent N volatilization (based on N:P ratio analysis) increased 29%. During the present study, CP in the cattle diet increased from approximately 13.5 to 15% in April 2003 with the addition of higher N content corn gluten feed to the ration. This diet change increased average fed N by approximately 24 g head d⁻¹ (15%) between summer 2002 and summer 2003. NH₃–N loss increased by 13 g head d⁻¹ (10%) during the same interval, accounting for 54% of the fed N increase. The increase in fed N and emissions was greater between winter 2003 and winter 2004. Winter fed N increased by 41 g head d⁻¹ (26%), and NH₃–N loss increased by 33 g head d⁻¹ (64%), so that the increase of NH₃–N loss was 80% of the fed N increase.

A study concurrent with this one was conducted during summer 2004 and spring 2005 by Flesch et al. (2007). They independently measured within-feedyard ammonia concentration using an open path laser, and used a 3-d sonic anemometer to measure wind speed and direction, atmospheric stability and turbulence statistics. These were used as inputs for the same BLS model used in this study. While some disagreement between the two studies is to be expected (different measurement locations mean the two studies “look” at emissions from different areas of the feedyard), their results agreed closely (Table 4). Mean ammonia emission rates of the two studies were within 7% of each other in 2004 and within 5% in 2005. However, the studies had only 7 days of common data. When mean ammonia emission rates of the common days were compared, the two studies agreed within 4% of each other. Close agreement in summer 2004 is not surprising, given that both methods provided continuous ammonia concentration as input to the BLS model. In spring 2005, however,

this study used gas washing to measure ammonia concentration, on 2.5-h time steps collected five times a day. Agreement with the results of Flesch et al. (2007), which used more detailed data, suggests that the BLS model is fairly robust as long as it is provided good quality data, and that calculations are relatively insensitive to the length of the sampling interval (Flesch et al., 2007, used 15-min intervals).

Annualized NH₃–N emission rate, calculated as the mean of summer and winter emissions, was 4430 kg NH₃–N d⁻¹, which was 53% of N fed to cattle. Emissions from the spring 2005 trial were not included in the annualized emission rate because of a limited number of days. However, with the expectation that the spring mean emission rate would be intermediate between that of summer and winter, the spring emission rate of 4770 kg NH₃–N d⁻¹ was within 8% of the annualized mean emission rate.

3.2. Emission factors

Emission factors were calculated on the basis of (i) the total number of cattle produced in one year by the feedyard, and (ii) the total biomass produced in 1 year, estimated from the start of feeding to slaughter for each head produced. We estimated the annual capacity, or total production, of the feedyard was 100 465 head, based on a mean one-time capacity of 44 651 head and 2.25 turnovers per year, typical of southern High Plains feedyards (Table 5). Biomass production was estimated to be 275 kg head⁻¹, based on an average starting weight of 275 kg head⁻¹ and a final slaughter weight of 550 kg head⁻¹. The emission factors for the pen area of this feedyard were 19.5 kg NH₃ (head fed)⁻¹ or 71.0 kg NH₃ Mg⁻¹ biomass produced.

Flesch et al. (2007) quantified ammonia emissions from the adjacent retention pond using an open path laser and sonic anemometer to measure inputs for a BLS model.

Table 4

Ammonia–N emissions from this study and the independent, concurrent study of Flesch et al. (2007)

	BLS model inputs	
	This study ^a	Flesch et al. (2007) ^b
Summer 2004		
No. of days ^c	12	12
NH ₃ emission rate (kg d ⁻¹)	7810	7300
Per capita NH ₃ –N emission rate (g head ⁻¹ d ⁻¹)	131	123
NH ₃ –N as % of fed N	64	63
Spring 2005		
No. days ^d	4	10
NH ₃ emission rate (kg d ⁻¹)	5800	6100
Per capita NH ₃ –N emission rate (g head ⁻¹ d ⁻¹)	118	124
NH ₃ –N as % of fed N	62	65

^a Gas washing/wet chemistry or chemiluminescence, wind and temperature profiles.

^b Open path laser, 3-d sonic anemometer.

^c 4 Days in common.

^d 3 Days in common.

Table 5

Annual production, ammonia emission and emission factors for feedyard pens and retention pond

Production ^a (head yr ⁻¹)	100 465
Total biomass produced ^b (Mg yr ⁻¹)	27 628
NH ₃ emission rate (Mg yr ⁻¹)	1962
NH ₃ emission factor, pens (kg NH ₃ [head fed] ⁻¹)	19.5
NH ₃ emission factor, pond (kg NH ₃ [head fed] ⁻¹)	0.7
NH ₃ emission factor, pens (kg NH ₃ Mg ⁻¹ biomass produced)	71.0
NH ₃ emission factor, pond (kg NH ₃ Mg ⁻¹ biomass produced)	2.4

^a Based on mean one-time capacity of 44 651 head and 2.25 turnovers per year.

^b Based on average starting weight of 275 kg head⁻¹ and final slaughter weight of 550 kg head⁻¹, giving total feedyard biomass production of 275 kg head⁻¹.

Retention pond ammonia emissions were 2.3% of pen ammonia emissions over 12 d in summer 2004 and 4.5% over 10 d in spring 2005, and varied because of different pond surface areas. Assuming retention pond emissions are 3.375% of pen emissions (mean of 4.5% in summer and 2.25% in winter) adds 66 Mg NH₃ yr⁻¹ to the annual emission rate we found, which increases the emission factor (for pens and pond) to 20.2 kg NH₃ (head fed)⁻¹, or 73.4 kg NH₃ Mg⁻¹ biomass produced (Table 5).

Previously reported or compiled emission factors for fed beef cattle were quite variable or based on limited data. Some of the first compiled emission factors were primarily based on European production systems (Asman, 1992; Battye et al., 1994), and ranged from 1.6 to 13.04 kg NH₃ head⁻¹ yr⁻¹ (Table 6). The USEPA (2004) based its emission factor for drylot beef and heifers (11.4 kg NH₃ head⁻¹ yr⁻¹) on two studies with limited data. In contrast, the emission factor of 19.5 kg NH₃ head⁻¹ yr⁻¹ for 275–550 kg beef steers and heifers housed in open lot pens that we report here is based on extensive data from 39 days of measurement taken during 5 months over 3 years.

The amount of protein fed to cattle has a major effect on ammonia emissions and must be considered (Fig. 4). Optimal CP for beef cattle diets is about 13% (Gleghorn

et al., 2004), and is greater during early feeding and less as cattle approach final weight. Todd et al. (2006) reported that reducing CP from 13 to 11.5% late in the finishing period, which closely matched the physiological requirements of the finishing steers near slaughter weight, decreased ammonia emission by 28%. Cole et al. (2006) observed a 22% decrease in apparent N volatilization (based on N:P ratio analysis) when CP was similarly reduced late in the finishing period. Diets fed during this study, with 13.5–15% CP, provided excess nitrogen, and most excess nitrogen is excreted as urinary urea and lost as ammonia. We speculate that fine-tuning the diets fed during this study to more closely match protein requirements of cattle could reduce the emission factor by 20–30%, to a range of 13.6–15.6 kg NH₃ head⁻¹ yr⁻¹.

Our estimated emission factor for the retention pond (0.7 kg NH₃ head⁻¹ yr⁻¹) is liberally based on the experimentally determined values of pond emissions reported by Flesch et al. (2007), and is about 2.4% of fed N. The USEPA (2004) reported storage pond emissions as 71% of N input to a pond. We estimate that for the feedyard studied here, about 5% of fed N runs off to the retention pond (Gilbertson et al., 1970; Bierman et al., 1999). If 71% of that N input is lost as ammonia-N, then ammonia-N loss from the retention pond is about 3.5% of fed N, which is reasonably close to our estimate (2.4%), especially considering the uncertainties involved in calculating a feedyard N balance. Ammonia emission from retention ponds may be highly variable because it depends on factors such as runoff, pond chemistry and surface area, but it will most likely be a very small percentage of nitrogen fed to cattle.

4. Conclusions

Ammonia emission rates and emission factors for a commercial beef cattle feedyard on the southern High Plains were quantified using measured profiles of ammonia concentration, wind speed and air temperature, and an inverse dispersion model. Data were collected on 39 days during 5 months over 3 years. Mean summer emission rate was 7420 kg NH₃ d⁻¹, and winter emission rate was about half that, at 3330 kg NH₃ d⁻¹. Annualized NH₃-N emission rate was 4430 kg NH₃-N d⁻¹, which was 53% of

Table 6

Comparison of ammonia emission factors for beef cattle production systems

Study	Ammonia source area	Animal type	Emission factor (kg NH ₃ head ⁻¹ yr ⁻¹)
This study	Open lot pens	Beef steers and heifers, 275–550 kg	19.5
	Retention pond	Beef steers and heifers, 275–550 kg	0.7
USEPA (2004)	Drylot	Beef and heifers	11.4
	Storage pond	Beef and heifers	71% of N input to pond
Battye et al. (1994)		Heifers > 227 kg	13.04
		Steers > 227 kg	8.22
Asman (1992)	Stable + storage	Fattening calves	1.6
		Young cattle for fattening	5.76
Misenheimer et al. (1987)	Beef cattle feedlots		5.9

the N fed to cattle. Emission rates agreed closely with those found in an independent, concurrent study. Daily per capita $\text{NH}_3\text{-N}$ losses increased by 10–64% after the dietary N content increased by 15–26%. Annual emission factors for the pen area of the feedyard were $19.5 \text{ kg NH}_3 (\text{head fed})^{-1}$, or $71.0 \text{ kg NH}_3 \text{ Mg}^{-1}$ biomass produced. Though not measured in this study, a best-estimate of annual emission factors for the retention pond of the feedyard was $0.7 \text{ kg NH}_3 (\text{head fed})^{-1}$, or $2.4 \text{ kg NH}_3 \text{ Mg}^{-1}$ biomass produced.

We found a general agreement in ammonia loss from beef cattle feedyards among studies conducted on the High Plains during the last 8 years. Annual ammonia loss tends to be about 50% of fed nitrogen. Summer emissions are about twice as great as in the winter. Ammonia emission is sensitive to crude protein content of cattle diets, and increases as protein increases beyond cattle requirements.

Emission factors from this study are probably greater than those from a feedyard with more typical diets with crude protein around 13%. However, higher protein feeds like corn gluten feed and distillers grains could become more common components of rations if more corn is diverted to processes such as wet milling and ethanol production. Higher nitrogen diets will result in greater ammonia emissions and will increase the challenge to reduce the amount of fugitive ammonia released to the atmosphere.

This research greatly expanded the database of ammonia emissions from beef cattle feedyards. However, longer term monitoring of ammonia emissions from feedyards is needed, over a greater range of management practices, such as diets, manure harvesting, and sprinkler dust control. Inverse dispersion models, such as the BLS model used here, show great utility and could be useful in a wide variety of monitoring and simulation applications.

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