Ammonia Emissions from Open Lot Beef Cattle Feedyards on the Southern High Plains

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ABSTRACT

Human activity has more than doubled the amount of reactive nitrogen that cycles through terrestrial ecosystems, with many negative impacts on ecosystem function and health, and air quality. Concentrated animal feeding operations (CAFO) are major sources of ammonia emitted to the atmosphere. There is a 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 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 five months over three years. Mean summer emission rate was 7420 kg NH_3 d^{-1}, and winter emission rate was about half that, at 3330 kg NH_3 d^{-1}. Annual NH_3-N emission rate was 4430 kg NH_3-N d^{-1}, which was 53% of the N fed to cattle. Daily per capita NH_3-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 NH_3 (head fed)$^{-1}$, or 70.2 kg NH_3 Mg^{-1} biomass produced. Annual emission factors for the retention pond of the feedyard were estimated to be 0.9 kg NH_3 (head fed)$^{-1}$, or 3.2 kg NH_3 Mg^{-1} biomass produced.

INTRODUCTION

Human activity has more than doubled the amount of reactive nitrogen that cycles through terrestrial ecosystems[1,2], with many negative impacts on ecosystem function and health, and air quality. Estimates of the contribution of agriculture to reactive N in the environment [3,4,5,6] range from 50 to >90%, with animal agriculture contributing the majority.

Concentrated animal feeding operations (CAFO) are major sources of ammonia emitted to the atmosphere. There is a considerable literature on ammonia emissions from poultry and swine CAFO, but few comprehensive studies have investigated large, open lot beef cattle feedyards. Hutchinson[7] was one of the first to quantify ammonia emissions from a commercial feedyard. Researchers in Nebraska[8,9,10] used a mass balance approach to quantify N at various points in the feedyard system, and calculated ammonia emission as the residual of the N balance. Researchers in Texas employed micrometeorological methods such as the flux-gradient method[11] or an inverse dispersion model[12] to quantify ammonia emissions from a Texas cattle feedyard.

The Texas Panhandle is the center of the southern High Plains cattle feeding industry. Over 7 million head of cattle, about a third of the US total, are fed annually within 250 km of Amarillo, TX. 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[13]. 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 with 13-13.5% crude protein (CP), often supplemented with urea[14].

There are two main sources of ammonia on feedyard surfaces; NH_4^+ hydrolized 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[15,16,17]. Most excreted nitrogen is in urine, ranging from 60-80%[18], and typically increases as crude protein in a diet increases beyond the physiological needs of animals[9,14,19,20]. In contrast, mineralization is a much slower process and provides a more constant, slow rate source of NH_4^+.

Micrometeorological methods 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[21,22,23]. Generally speaking, micrometeorological methods rely on measurements in and characterization of the atmosphere near the ground. They have rarely been used to characterize ammonia emissions from beef cattle feedyards[7,12,23,24]. Quantifying ammonia, or any other gaseous emisssions, from beef cattle feedyards entails two major challenges: i) measurement of atmospheric NH_3; ii) an estimate of NH_3 emission from a surface to the atmosphere based on direct measurement or on a flux model that describes or simulates the turbulent dispersion of gases.

Complex dispersion models are based on the description of the relationship between a source of a gas and a downwind receptor or point[25]. Assumptions about turbulent flow are made to establish the relationship[26]. Sometimes, source strength of a gas is known, and a dispersion model predicts concentration at the receptor. A Gaussian plume model is an example of this type of dispersion model, in which empirical parameters describe the three-dimensional spread of a plume of gas from its source. The backward Lagrangian stochastic (BLS) model estimates flux of a gas by taking concentration of a gas measured at a point, say downwind of an emitting source, and modeling the trajectories of thousands of gas particles backward to the emitting source[27]. Advantages of the BLS model include a small number of required inputs (gas concentration, wind speed and direction, atmospheric stability, defined source area). The BLS model has been tested and compared positively to other methods for estimating fluxes of methane[28], ammonia[29], and with gas release experiments[26].

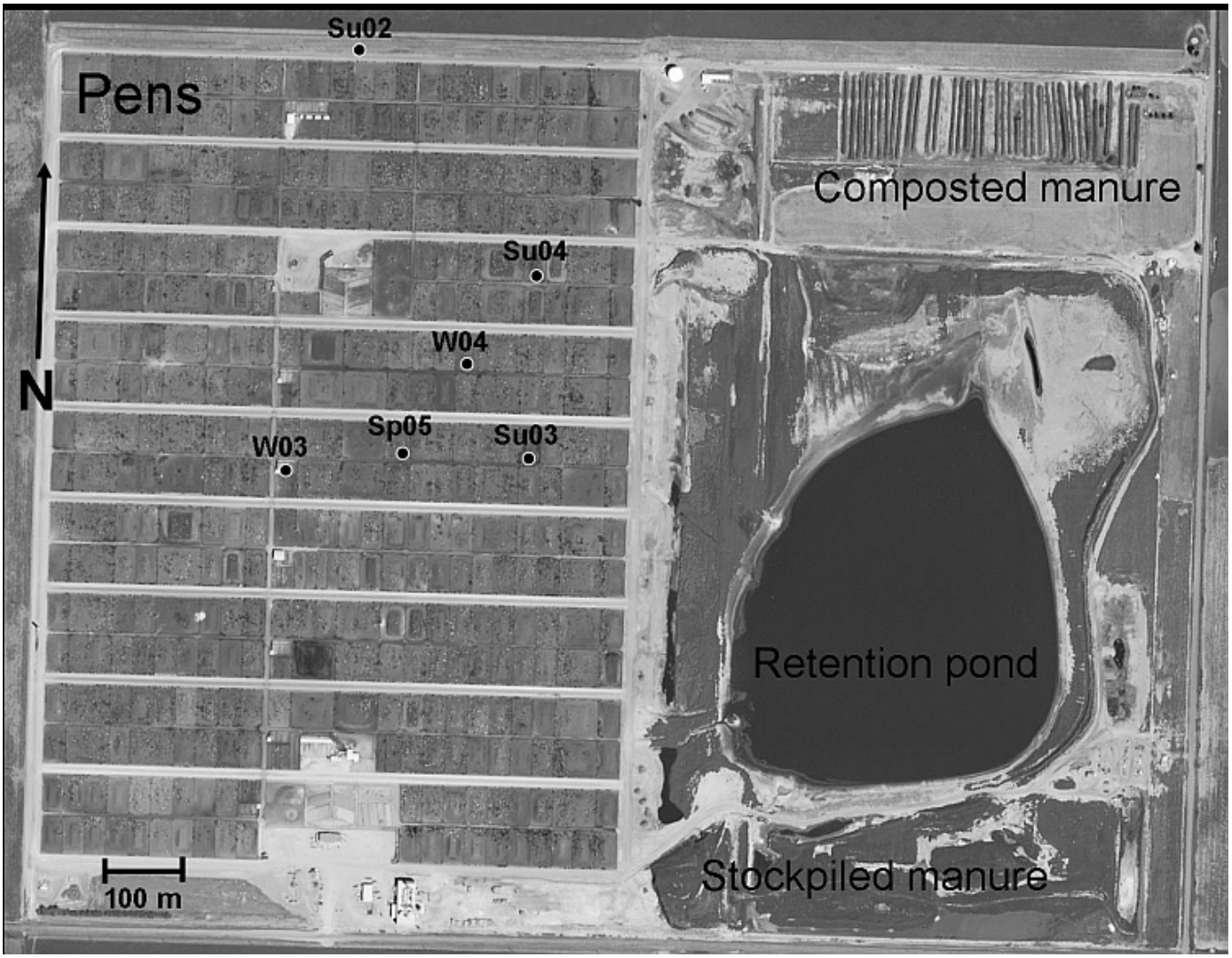
Our objectives were to i) measure atmospheric ammonia, wind speed and temperature profiles at a typical commercial beef cattle feedyard for extended time periods in summer and winter; ii) use these measurements as inputs into a BLS dispersion model to quantify ammonia emissions; and iii) calculate ammonia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

MATERIALS AND METHODS

Location And Site Description

Research was conducted at a commercial beef cattle feedyard located in the Texas Panhandle (Figure 1). Mean occupancy of the 77-ha pens was 44 651 head, with an inverse stocking density of 17 m^2 $head^{-1}$. Median capacity of feedyards in the region is 30 000 head. 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 mobile cattle. A retention pond, manure stockpiles and compost ricks were located east of the pens. Retention pond area was variable, depending on precipitation and runoff from pens, and ranged from 20-25 ha. 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

Figure 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.



rapidly evaporates. Prevailing winds are southerly to southwesterly, with wind direction almost half the time between 160° and 250°.

Six field campaigns were conducted, commencing in summer 2002 and ending in spring 2005 (Table 1). During each campaign, an instrument tower (Figure 2) 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 prevailing wind directions, neccessities for power, and feedyard logistics.

Tower Measurements

Ammonia concentration was measured using acid gas washing sampled on the tower at different heights for each campaign, but sampling protocol was the same (Table 1). Ammonia was trapped in gas

Figure 2. Tower erected in feedyard pen, summer 2003, with instruments used to measure air temperature, wind speed, and ammonia concentration profiles. A similar tower was used during all campaigns, though measurement heights varied (see Table 1).

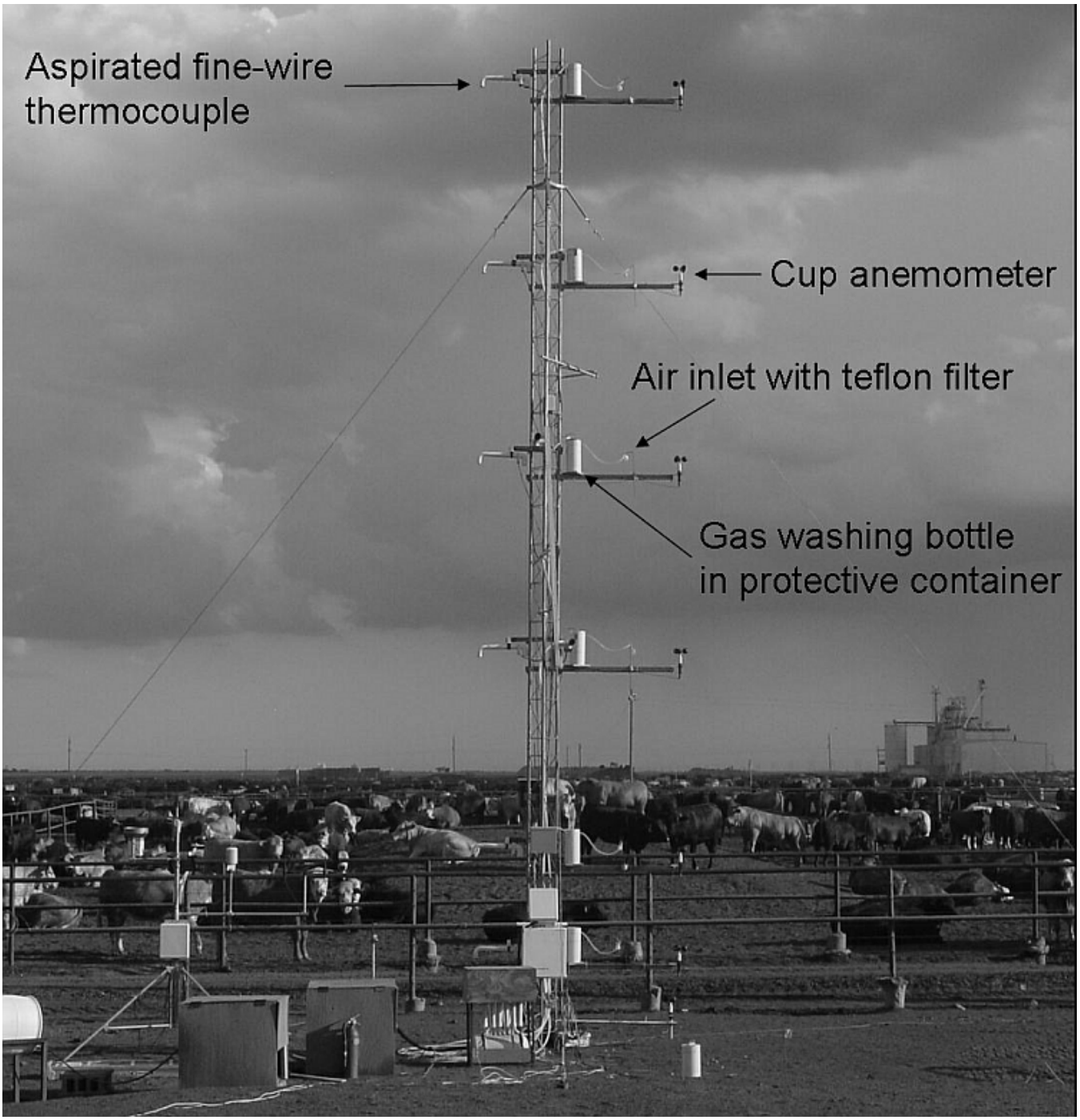


Table 1. Field campaign dates, sample integration times for ammonia concentration measurements, and profile heights for ammonia concentration, wind speed and air temperature.

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

[†] Sample time was 2 h out of every 4 h for 12 h during daytime; 2 h from 0000-0200 h during nighttime.
[‡] Samples were collected every 2.5 h for 10 h during daytime; 2.5 h from 0000-0230 h during nighttime.

washing bottles by first drawing air through a teflon filter to remove particulates, then bubbling it through an impinger in 80 to 120 ml of 0.1 N H_2SO_4. Air flow rate of each gas washing bottle was measured with a precision, calibrated flow meter (Dry-Cal DC Lite, Bios International, Butler, NJ) 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 the 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 in the samples, with a minimum detection limit of about 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 m and 6 m was measured sequentially using a 3-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[11].

Profiles of wind speed and air temperature were defined 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.

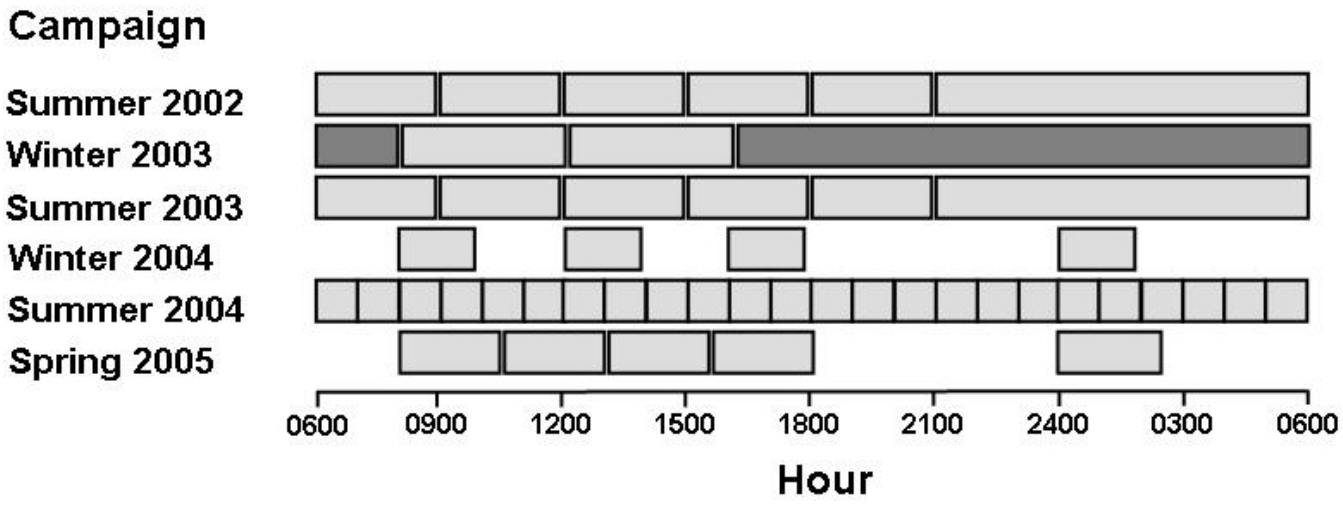
BLS Model

The backward Lagrangian stochastic dispersion model used to estimate emissions was Windtrax (Thunder Beach Scientific, Halifax, Canada), a commerically available model. Details of the model's theory, development and testing were given in Flesch et al.[26,29].

Ideally, inputs to the BLS model should have sample integration times of 15–120 min[26,29] in order to meet the assumption of stationarity. Gas washing requires longer sample integration times. Sommer et al.[28] pointed out this problem, especially with regard to atmospheric stability. They used passive ammonia samplers, with concentration integrated over time periods of 5 to 26 h and found that BLS flux estimates were within 16-24% of integrated horizonal flux estimates. They recommended the assignment of neutral stability for longer sample integration times that may violate the assumption of stationarity.

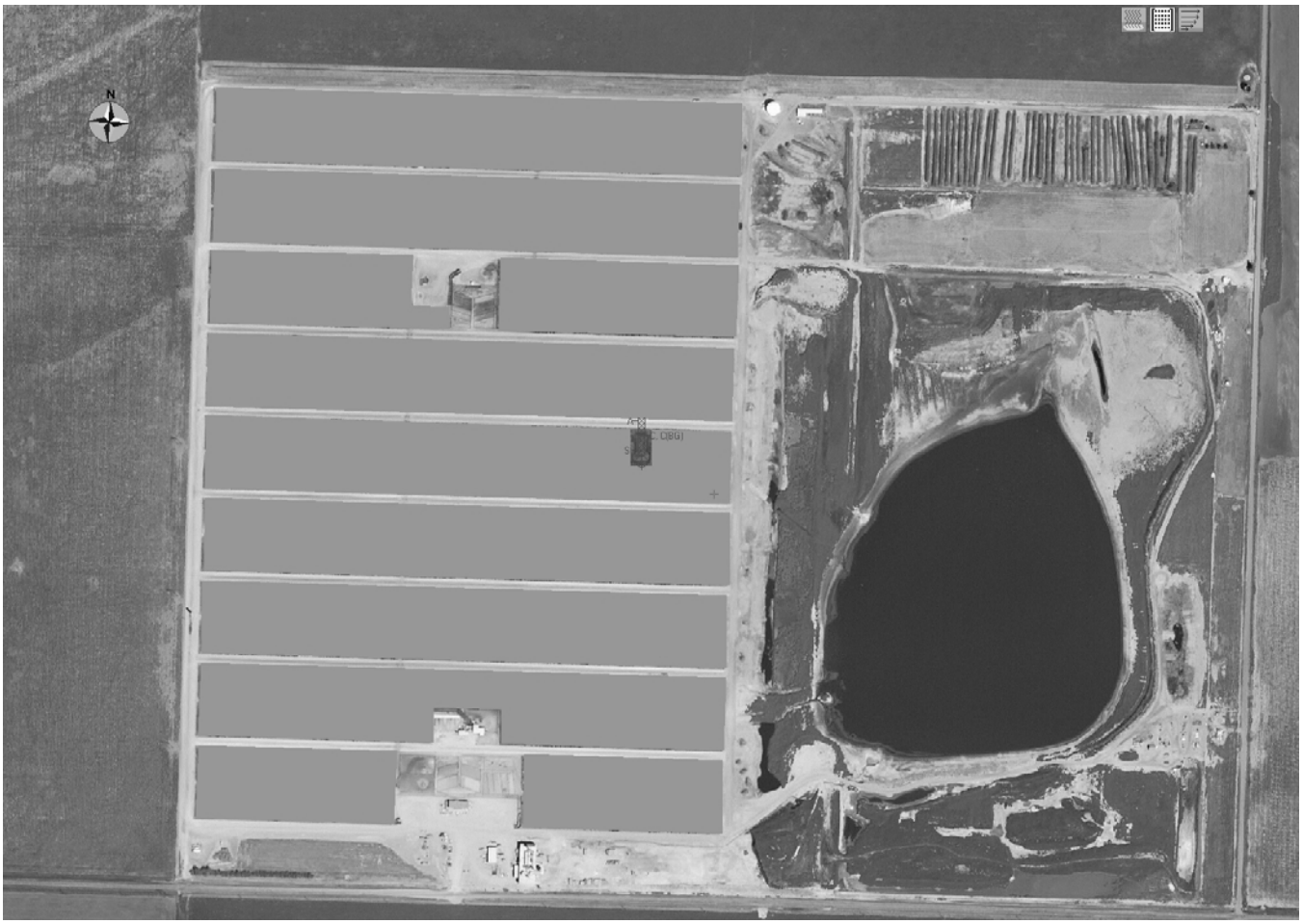
The ammonnia concentration sampling schedules for each campaign are shown in Figure 3. For the first three campaigns, we adopted either 3-h (summers 2002 and 2003) or 4-h (winter 2003) daytime sample times. We found that in most cases, the calculated nighttime (over 9 h in summer or 16 hours in winter) stability was near neutral. Daily mean emission for these three campaigns was the time-weighted mean 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.

Figure 3. Sample integration times for ammonia concentration measurements. The darker gray for Winter 2003 indicates that the nighttime sampling period was continuous from 1600 to 0800 the following day. Ammonia concentration was measured using gas washing, except for the 1-h continuous measurements of Summer 2004 that were made using chemiluminescence.



The feedyard pens source area was mapped into Windtrax as polygons defined by GPS coordinates (Figure 4). Pens were defined as manure surfaces either with cattle occupancy or recently occupied. Service roads and feed truck alleys between pens were excluded from the source map. The

Figure 4. Windtrax source map for the summer 2003 campaign, with source polygons (gray rectangles) defined from GPS coordinates. The mapped tower, with model input specifications, is visible near the right center of the pens, located in a vacant pen that was excluded as a source area.



vacant pen in which the tower was located was also excluded as a source. 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.

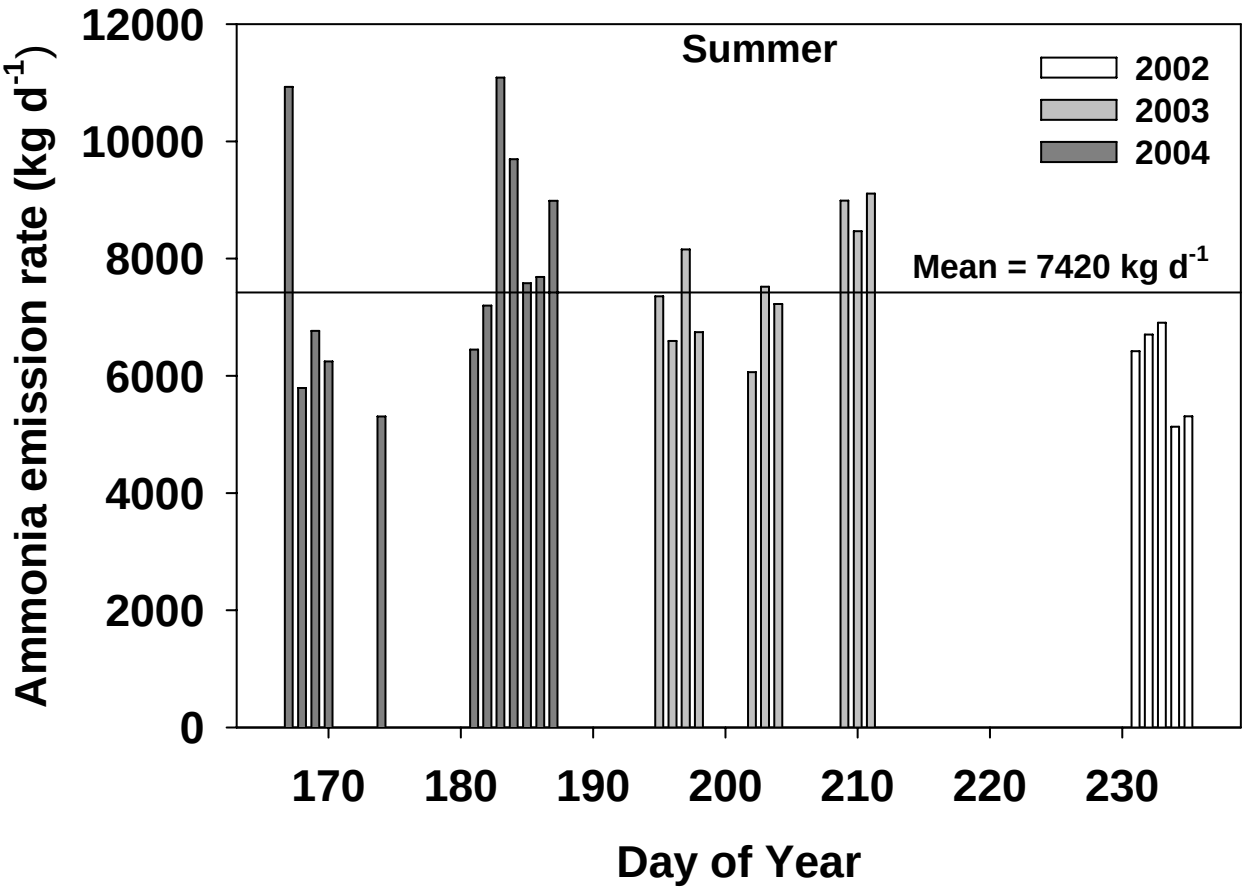
A BLS simulation for each sampling period was run at each measurement height, using measurements of wind speed and ammonia concentration from that height. Other inputs were wind direction and Monin-Obukhov length (MOL). A single MOL was calculated from measurements of wind speed and air temperature profiles: first the gradient Richardson number[30] was calculated; then, semi-empirical expressions relating MOL to Richardson number were applied[31]. An ensemble of 50 000 particles was used for each simulation. Roughness length (z_0 = 0.10 m) was determined from sonic anemometer measurements at the feedyard. The emission rates from each height were averaged to give the emission rate for a sampling period. Daily emission rate was calculated, as explained above, by either calculating the time-weighted mean of the emisson rates for a day's sampling periods, or integrated using the trapezoidal rule.

RESULTS AND DISCUSSION

Ammonia Emission Rates

During summer campaigns (June, July, August), with a total of 27 days, daily mean ammonia emission rate ranged from 3820 to 10 930 kg d^{-1} (Figure 5). Mean ammonia emission rate (± standard

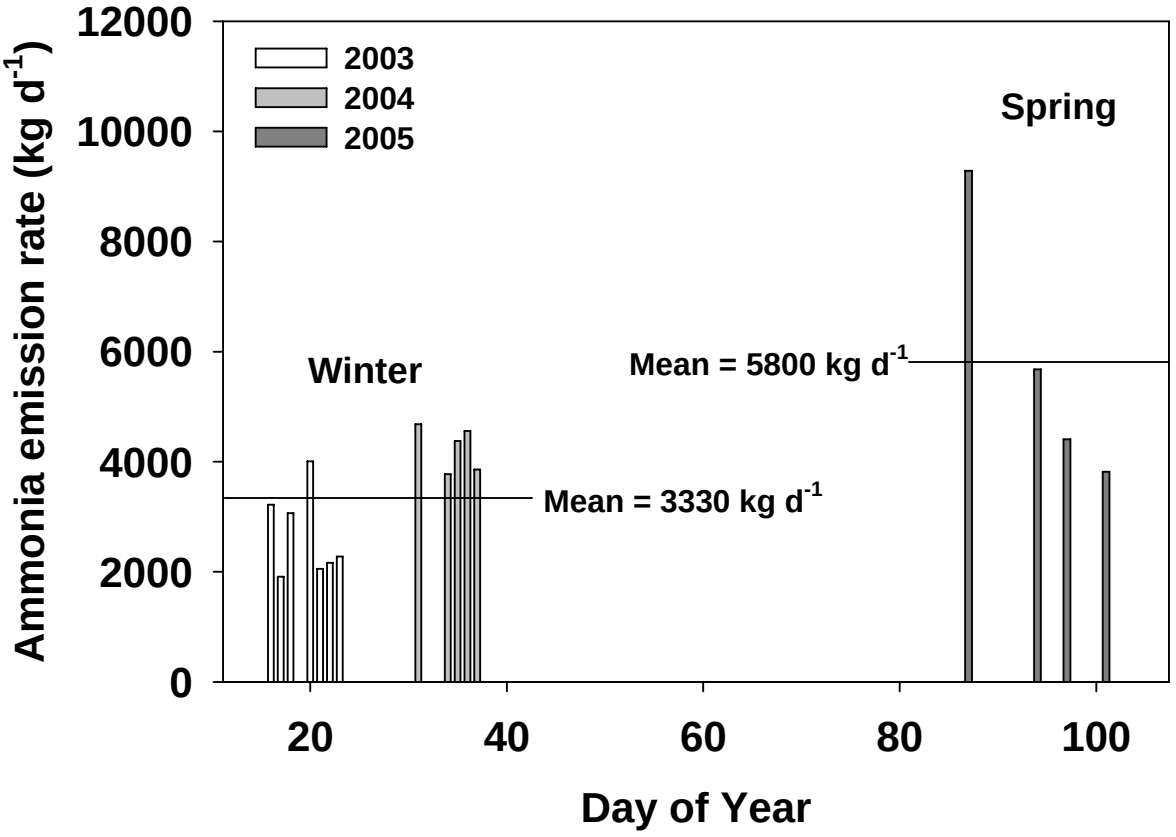
Figure 5. Daily mean ammonia emission rate during summer for three campaigns conducted in 2002, 2003 and 2004.



deviation) during summer was 7420 ± 1580 kg d^{-1}. Ammonia emission during winter (January, February, 12 days) ranged from 1910 to 4680 kg d^{-1} (Figure 6). Mean ammonia emission rate 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} (Figure 6).

Daily per capita NH_3-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; Figure 7). During winter, one-time capacity per capita NH_3-N loss decreased, compared with summer

Figure 6. Daily mean ammonia emission rate during winter and spring for three campaigns conducted in 2003, 2004 and 2005.



losses, to 51 g head d^{-1} in 2003 and 84 g head d^{-1} in 2004; mean winter loss was 64 ± 21 g $head^{-1}$ d^{-1} (n=12). One-time capacity per capita NH_3-N loss in spring 2005 averaged 118 ± 50 g head d^{-1} (n=4).

Ammonia-N loss during summer was 72%, 70% and 64% of fed N, in 2002, 2003 and 2004, respectively; mean summer NH_3-N loss was 68 ± 13% of fed N (Table 2). As a percentage of fed N, wintertime NH_3-N loss was 32% in 2003 and 42% in 2004, and averaged 36 ± 9% of fed N. Ammonia-N loss in spring 2005 was 62 ± 26% of fed N. These values compare favorably to those reported in research from Texas, Nebraska and New Mexico (Table 3). The Nebraska work[8,9,10] was based on quantifying the feedyard N balance. Ammonia-N loss, as the residual of the N balance, ranged from 51–63% during summer, and was 35% in winter. Texas research used methodology similar to this study[12,32]. Summertime ammonia-N loss ranged from 53–63%, and winter loss was 29%. An estimate of N volatilization loss is provided by using the nitrogen:phosphorus ratio in feed and manure[24]. The N:P ratio of feedyard pen manure is less than the N:P ratio of feed because N is reduced by retention in animals and by loss as gaseous N (e.g. as NH_3, N_2O or N_2), and because P is conservative. Using this method, Todd et al.[24] found that 45% of fed N was lost as gaseous N. Using the N:P ratio method, Cole et al.[39] found that gaseous N loss ranged from 51-65% 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%.

Table 2. Per capita nitrogen fed to cattle, per capita ammonia-N loss, and percentage of fed nitrogen lost as ammonia-N.

Campaign	No. days	Per capita fed N	Per capita NH_3-N loss	NH_3-N loss as % of fed N
		---------- g $head^{-1}$ d^{-1} ----------		%
Summer 2002	5	162	117	72
Winter 2003	7	160	51	32
Summer 2003	10	186	130	70
Winter 2004	5	201	84	42
Summer 2004	12	205	131	64
Spring 2005	4	193	118	62

Figure 7. Mean daily per capita fed N and ammonia-N losses, by campaign (summer, Su, winter, W, and spring, Sp, followed by year).

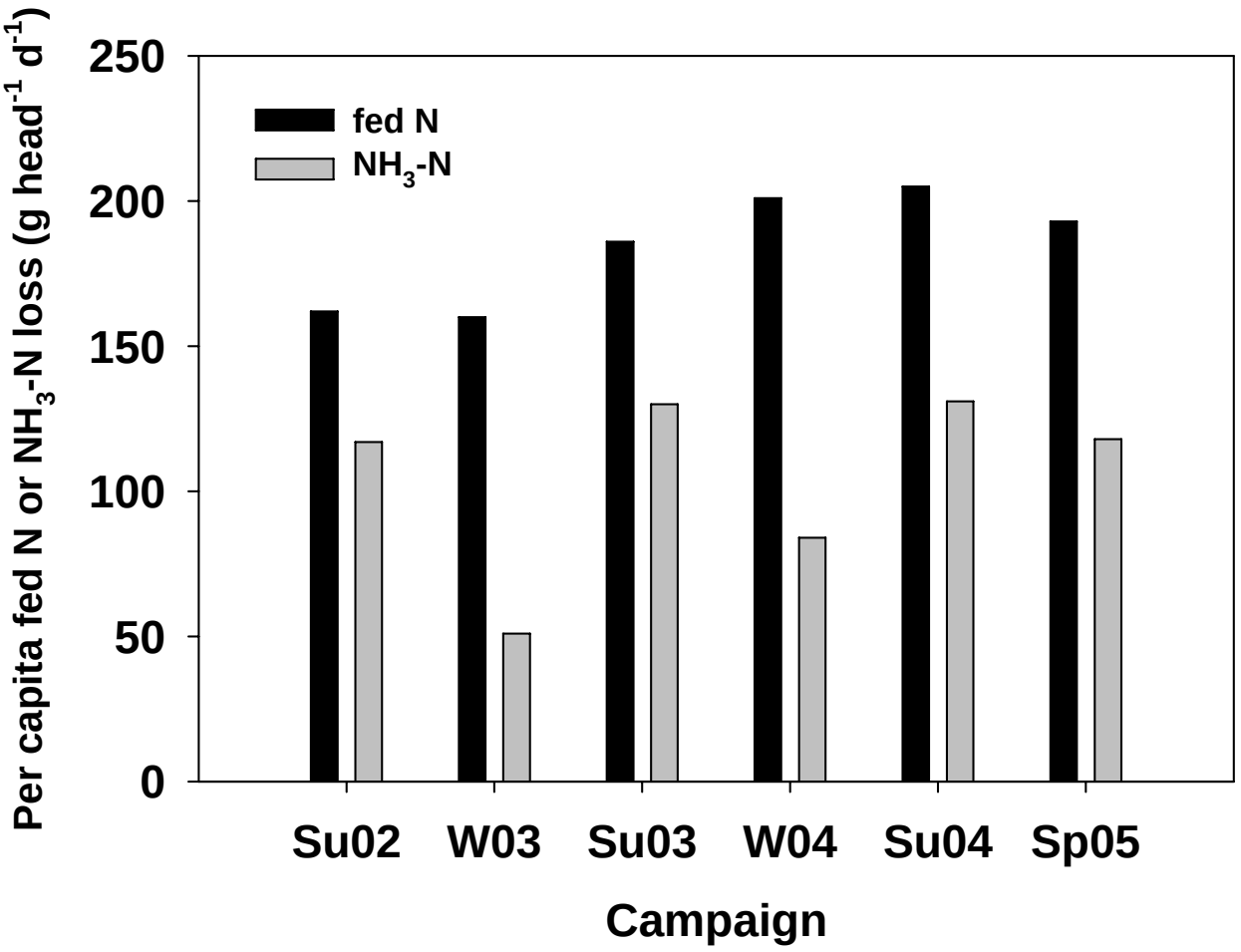


Table 3. Comparison of ammonia-N loss as a percentage of fed nitrogen from High Plains beef cattle feedyards.

Study	Summer	Winter	Annual
	---------- % ----------		
This study[†]	68	36	53
Flesch et al.[12 †]	63		
Harper et al.[32 †]	53	29	
Todd et al.[24 † ¶]	45	44	45
Cole et al.[39 § ¶]	51-65		
Erickson and Klopfenstein[10‡]	51-61		
Erickson et al.[9 ‡]	63		
Bierman et al.[8 ‡]	53-63	35	

† Texas
‡ Nebraska
§ New Mexico
¶ Gaseous N loss, based on change in N:P ratio of feed and manure.

From 30 to 50% of N in cattle diets is routinely excreted as urinary N. This value increases when fed N exceeds animal requirements, and can contribute to ammonia volatilization. For example, Todd et al.[18] found that increasing dietary crude protein (CP) from 11.5% to 13% increased ammonia emission from an artificial feedyard surface by 39%. Cole et al.[39] found that when CP increased from 11.5% to 13%, apparent N volatilization (based on N:P ratio analysis) increased 29%. During this study, crude protein in the cattle diet increased from approximately 13.5% to 15% in April 2003 with the addition of corn gluten feed to the ration. This diet change increased fed N by 24 g head d^{-1} (15%) between summer 2002 and summer 2003. NH_3-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%), while NH_3-N loss increased by 33 g head d^{-1} (64%), so that the increase of NH_3-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.[12]. They 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. Results from the studies 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 seven 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.[12], which used more detailed data, suggests that the BLS model is fairly robust as long as it is provided good quality data.

Table 4. Comparison of ammonia-N emissions from this study and the independent, concurrent study of Flesch et al.[12].

	BLS model inputs	
	This study[†]	Flesch et al.[12‡]
Summer 2004		
No. days[§]	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[¶]	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

[†] Gas washing/wet chemistry or chemiluminescence, wind and temperature profiles.
[‡] Open path laser, 3-d sonic anemometer.
[§] Four days in common.
[¶] Three days in common.

Annualized NH_3-N emission rate, calculated as the mean of summer and winter emissions, was 4430 kg NH_3-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 NH_3-N d^{-1} was within 8% of the annualized mean emission rate.

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 one year from the start of feeding to slaughter. 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 19.3 kg NH_3 $(head\ fed)^{-1}$ or 70.2 kg NH_3 Mg^{-1} biomass produced.

Flesch et al.[12] quantified ammonia emissions from the adjacent retention pond using an open path laser and sonic anemometer to measure inputs for a BLS model. 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 a maximum 4.5% of pen emissions adds 87 Mg NH_3 yr^{-1} to the annual emission rate we found, which increases the

emission factor (for pens and pond) to 20.2 kg NH_3 (head fed)$^{-1}$, or 73.4 kg NH_3 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[33,34], and ranged from 1.6 to 13.04 kg NH_3 head^{-1} yr^{-1} (Table 6). USEPA[35] based its

Table 5. Annual production, ammonia emission and emission factors for feedyard pens and retention pond.

Production[†] (head yr^{-1})	100 465
Total biomass produced[‡] (Mg yr^{-1})	27 630
NH_3 emission rate (Mg yr^{-1})	1940
NH_3 emission factor, pens (kg NH_3 [head fed]$^{-1}$)	19.3
NH_3 emission factor, pond (kg NH_3 [head fed]$^{-1}$)	0.9
NH_3 emission factor, pens (kg NH_3 Mg^{-1} biomass produced)	70.2
NH_3 emission factor, pond (kg NH_3 Mg^{-1} biomass produced)	3.2

[†] Based on mean one-time capacity of 44 651 head and 2.25 turnovers per year.
[‡] 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}.

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	19.3
	retention pond	beef steers and heifers, 275-550 kg	0.9
USEPA[35]	drylot	beef and heifers	11.4
	storage pond	beef and heifers	71% of N input to pond
Battye et al.[34]		heifers > 227 kg	13.04
		steers > 227 kg	8.22
Asman[33]	stable + storage	fattening calves	1.6
		young cattle for fattening	5.76
Misenheimer et al.[38]	beef cattle feedlots		5.9

emission factor for drylot beef and heifers (11.4 kg NH_3 $head^{-1}$ yr^{-1}) on two studies with limited data. The emission factor of 19.3 kg NH_3 $head^{-1}$ 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 five months over three years.

The amount of protein fed to cattle has a major effect on ammonia emissions and must be considered (Figure 7). Optimal crude protein in beef cattle diets is about 13%[36], and is greater during early feeding and less as cattle approach final weight. Todd et al.[18] reported that reducing CP from an excessive 13% to 11.5%, which closely matched the physiological requirements of the finishing steers near slaughter weight, decreased ammonia emission by 28%. Cole et al.[39] observed a 22% decrease in apparent N volatilization (based on N:P ratio analysis) when CP was similarly reduced. Diets fed during this study, with 13.5-15% CP, provided excess nitrogen, and most excess nitrogen is excreted 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 13.5–15.4 kg NH_3 $head^{-1}$ yr^{-1}.

Our emission factor for the retention pond (0.9 kg NH_3 $head^{-1}$ yr^{-1}) is liberally based on the experimentally determined[12] value of pond emissions as 4.5% of pen emissions, which is about 2.4% of fed N. USEPA[35] 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[8,37]. 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 and pond chemstry and surface area, but it will most likely be a very small percentage of nitrogen fed to cattle.

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 five months over three years. Mean summer emission rate was 7420 kg NH_3 d^{-1}, and winter emission rate was about half that, at 3330 kg NH_3 d^{-1}. Annual NH_3-N emission rate was 4430 kg NH_3-N d^{-1}, which was 53% of the N fed to cattle. Emission rates agreed closely with those found in an independent, concurrent study. Daily per capita NH_3-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 NH_3 $(head\ fed)^{-1}$, or 70.2 kg NH_3 Mg^{-1} biomass produced. Annual emission factors for the retention pond of the feedyard were estimated to be 0.9 kg NH_3 $(head\ fed)^{-1}$, or 3.2 kg NH_3 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 eight 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 feedstuffs 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 emissons 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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