CONTINUOUS AMMONIA EMISSION MEASUREMENTS FROM A COMMERCIAL BEEF FEEDYARD IN TEXAS

M. B. Rhoades, D. B. Parker, N. A. Cole, R. W. Todd, E. A. Caraway,
B. W. Auvermann, D. R. Topliff, G. L. Schuster

ABSTRACT. *Ammonia emissions from cattle feedlots pose the potential to react with other compounds such as oxides of nitrogen and sulfur, which lead to detrimental environmental effects. Ambient ammonia (NH_3) concentrations were measured continuously at a beef cattle feedyard for 12 months beginning in March 2007. Concentrations were measured every 5 min, 24 hours per day, at a sample intake height of 3.3 m using a chemiluminescence analyzer. On-site weather data were collected concurrently. Modeled emissions of NH_3 were compared with the mass balance of N for the feedyard. Mean annual NH_3 concentrations were 0.57 ppm, with a monthly average low of 0.37 ppm in December 2007 and a monthly average high of 0.77 ppm in August 2007. Flux densities were calculated using a backward Lagrangian stochastic model (WindTrax 2.0.7.8). Mean annual flux density was 70.7 $\mu g\ m^{-2}\ s^{-1}$ (2.2 $kg\ m^{-2}\ year^{-1}$). Mean monthly flux density ranged from 42.7 to 123.1 $\mu g\ m^{-2}\ s^{-1}$ (0.11 to 0.32 $kg\ m^{-2}\ month^{-1}$) in November and April 2007, respectively. Both concentration and flux density had a diel distribution with minima during the nighttime hours and maxima during the early afternoon. On an annual basis, 48.8% of fed N was volatilized as NH_3. The inverse modeled daily ammonia production per head was 85.3 g NH_3-N (head fed)$^{-1}$ d^{-1}.*

Keywords. *Ammonia, Backward Lagrangian stochastic model, Beef feedyard, CAFO, Chemiluminescence, Emissions, WindTrax.*

The practice of concentrating large numbers of beef cattle into animal feeding operations (AFOs) has been well established in the U.S. Great Plains and many western U.S. states. These large-scale operations, coupled with supporting industries, have significant national and regional economic impact. The Texas Panhandle saw an economic impact of $7 billion in 2007 from cattle feeding alone (TCFA, 2008). After factoring in secondary industries and services, the total economic impact from the cattle feeding industry for the area is in excess of $19 billion. This streamlined movement of commodities and animals has proven an effective means of preparing animals for harvest and has been emulated in other countries. As commodities and animals are moved through the production system, nutrients associated with them are carried as well, and their fate and environmental impact need to be accounted for.

Most beef feedyards in the Southern Great Plains feed a corn-based diet, formulated to contain approximately 13.5% crude protein (CP, dry matter basis) (Vasconcelos and Galyean, 2007). However, several researchers have noted that decreasing the CP concentration in the diet can result in a decrease in the volatilization of ammonia (NH_3) from animal production facilities (Cole et al., 2005; Todd et al., 2006). Frank and Swensson (2002) found that NH_3 release from manure was reduced with lowered CP (from 17% to 13%) in the ration of dairy cows in Sweden. However, they found that while NH_3 emissions from manure were reduced, milk quality also tended to be reduced, although they cautioned that the form of the protein had a greater effect on milk quality than did overall CP percentage.

In a feedyard, NH_3 moves from the source area (i.e., the pen surface) to the atmosphere. While the atmosphere can be considered a source of NH_3 deposition when the atmospheric ammonia concentration is higher than the concentration at the feedyard surface, this rarely occurs and the feedyard is commonly considered to be the only source of ammonia emission. Ammonia flux, defined as the rate of ammonia volatilization per unit area of feedyard surface in units of mass area^{-1} time^{-1}, depends on several factors, including pen surface pH, pen surface and air temperatures, wind speed, moisture content of the source area, and nitrogen (N) concentration in the source (Duyson et al., 2003). Several researchers have established that most of the NH_3 in concentrated animal feeding operations (CAFOs) is

Submitted for review in June 2010 as manuscript number SW 8611; approved for publication by the Soil & Water Division of ASABE in September 2010.

The mention of trade names of commercial products in this article is solely for the purpose of providing specific information and does not imply recommendation or endorsement by the Texas A&M University System or the USDA.

The authors are **Marty B. Rhoades, ASABE Member Engineer,** Research Fellow, Department of Agricultural Sciences, West Texas A&M University, Canyon, Texas; **David B. Parker, ASABE Member Engineer,** Supervisory Agricultural Engineer, USDA-ARS U.S. Meat Animal Research Center, Clay Center, Nebraska; **N. Andy Cole,** Research Animal Scientist, and **Richard W. Todd,** Research Soil Scientist, USDA-ARS Conservation and Production Research Laboratory, Bushland, Texas; **Edward A. Caraway,** Research Associate, Department of Agricultural Sciences, West Texas A&M University, Canyon, Texas; **Brent W. Auvermann, ASABE Member Engineer,** Associate Professor, Texas Agri-Life Research, Amarillo, Texas; **Donald R. Topliff,** Associate Dean, Department of Agricultural Sciences, West Texas A&M University, Canyon, Texas; and **Greta L. Schuster,** Associate Professor, Department of Agronomy and Resource Sciences, Texas A&M University, Kingsville, Texas. **Corresponding author:** Marty B. Rhoades, Department of Agricultural Sciences, West Texas A&M University, WTAMU Box 60998, Canyon, TX 79016; phone: 806-651-2289; fax: 806-651-2504; e-mail: mrhoades@wtamu.edu.

volatilized from urine spots, as opposed to feces (Stewart, 1970; Ball et al., 1979; Hutchinson et al., 1982; Vallis et al., 1982; Whitehead and Raistrick, 1991; Harper et al., 2004; Cole et al., 2005; 2009; Koziel et al., 2005). Vallis et al. (1982) reported that within 2 h of urination up to 80% of the urea in urine can be hydrolyzed to ammonium, which is easily transformed to NH_3 and therefore readily available for volatilization.

The pathway for conversion of urea to NH_3 is presented in equations 1a to 1c (Hausinger, 2004):

$$CO(NH_2)_2 + H^+ + 2H_2O \xrightarrow{\text{UREASE}} HCO_3^- + 2NH_4^+ \quad (1a)$$

pH of 6.5 to 8.0

$$HCO_3^- + H^+ \rightarrow CO_2 + H_2O \quad (1b)$$

$$2NH_4^+ \leftrightarrow 2NH_3^- + 2H^+ \quad (1c)$$

As the pH increases, reaction 1c favors the release of NH_3, whereas at low pH (<6.5), most ammoniacal N is found in NH_4^+. As pH increases (>8), a molecule of urea, in the presence of water and urease, hydrolyzes rapidly into two molecules of NH_3.

Although the method and primary source of NH_3 volatilization for feedyards has been well established, there have been little data published on long-term concentration measurements and fluxes at beef cattle CAFOs. The objectives of this research were to:

- Quantify ambient NH_3 concentrations in a feedyard over an extended period of time.
- Estimate flux density based on local climate measurements using a backwards Lagrangian stochastic model.
- Calculate an NH_3 emission coefficient for a beef cattle CAFO pen surface.

Methods and Materials

Feedyard

The participating feedyard in this study has a one-time feeding capacity of 24,000 cattle and is located in the Texas Panhandle (fig. 1). Feedyard pen dimensions were 695 m × 420 m (pens, feed alleys, and drovers alleys inclusive). The runoff retention pond dimensions were 595 m × 77 m. This particular feedyard was selected based on its relative isolation from other CAFOS in the area and the rectangular geometry of the pen layout. No other CAFOs were located within 8 km upwind (in the direction of the predominate prevailing winds) of the feedyard. The per annum capacity of the feedyard, assuming a turnover of 2.25 times per year, would be 54,000 head. The feedyard population consisted of mostly crossbred steers and heifers with an average weight of 340 kg. Animal weights ranged from 225 kg for both steers

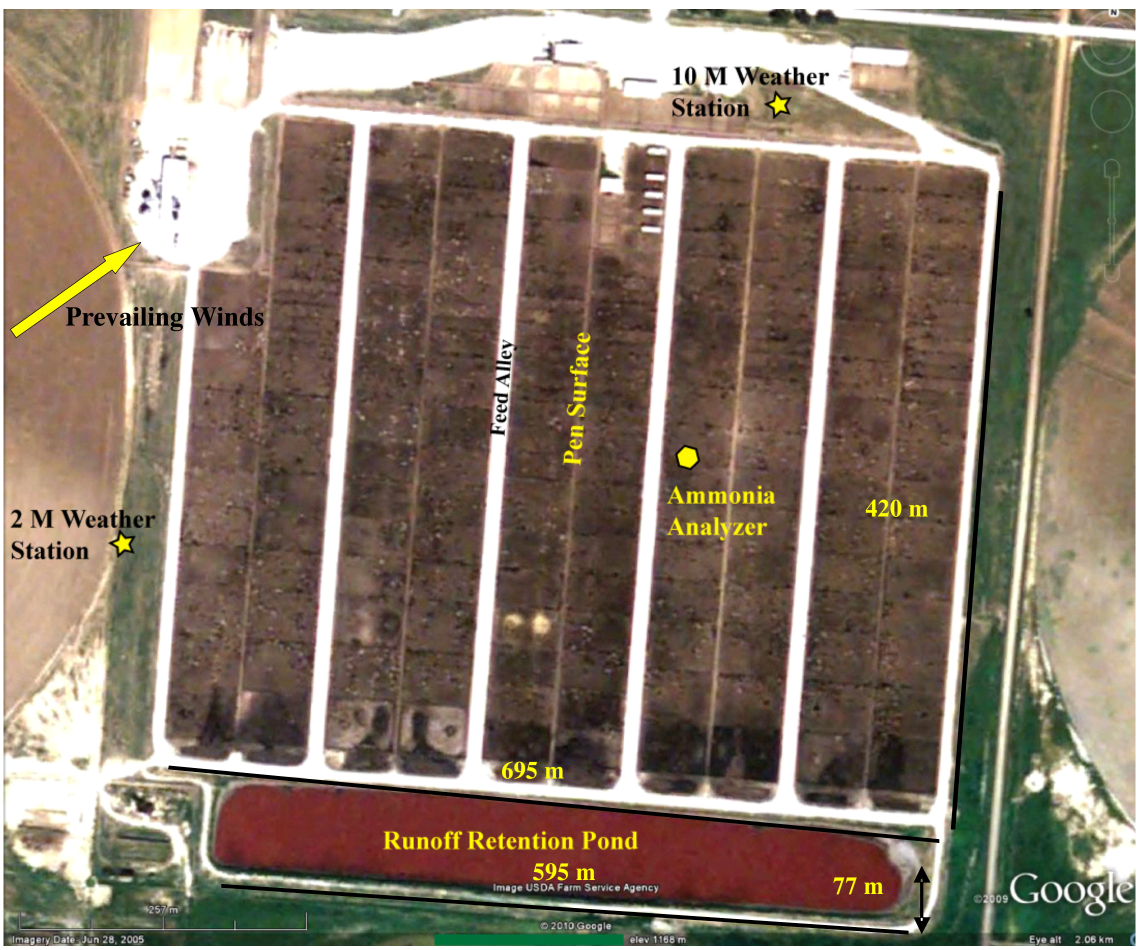

Figure 1. Satellite photograph of the participating feedyard showing equipment location. (Aerial photo from Google Earth, http://earth.google.com).

and heifers to 475 kg for heifers and 625 kg for steers. While not all ration information was available due to proprietary concerns, the feedyard in this study fed four different rations based on the weight and acclimation of the cattle on feed. Newly arrived cattle were fed ration 1, which had the greatest roughage to concentrate ratio, while finishing cattle were fed ration 4, which had the greatest concentrate to roughage ratio. Ration 4 comprised 90% of the fed diets, with rations 1, 2, and 3 equally split for the remainder. Diets were based on steam-flaked corn and corn silage. Crude protein concentrations in all rations varied slightly from month to month throughout the study period, but they averaged about 13% (DM basis) (data supplied by feedyard).

The feedyard harvested manure as requested by local farmers for land application. This occurred two to three times per year and was typical of the majority of commercial feedlots in the area. Manure harvesting consisted of scraping the manure pad down to the "hard pan," which was a densely compacted transitional layer between the manure and underlying soil. This scraped manure was temporarily stockpiled in the center of the pen. A front-end loader was then used to load manure into a spreader truck, which transported the manure to the field where it was applied.

Ammonia Sampling

Ammonia concentrations were monitored at a height of 3.3 m above the pen surface in the center of the feedyard pen area (fig. 1). While no background measurements were made at the location, ongoing work conducted by Auvermann (2009, unpublished) determined that the annual mean of atmospheric NH_3 for the geographic area to be about 0.0045 ppm. This was assumed to be the background concentration at the feedyard. Ammonia was measured continuously with a chemiluminescence analyzer (model 17C, Thermo Environmental Instruments, Franklin, Mass.) located inside a temperature-controlled instrument shelter.

The analyzer was calibrated weekly using instrument-grade air, certified standard span NH_3 gas in air (98 ppm, diluted to 4.66 ppm with instrument-grade air), and NO in nitrogen (50 ppm, diluted to 4.55 ppm with instrument-grade air) (AirGas Southwest, Amarillo, Tex.). The TEI 17C ammonia analyzer has a lower detectable limit of 0.001 ppm and accuracy of 1% of the full range setting. The instrument shelter was a modified 1.5 m × 2.1 m box trailer with a 3.9 kW air-conditioning unit. Data were recorded on a Campbell Scientific CR23X data logger using analog outputs from the analyzer. Ammonia concentrations were scanned every 10 s and averaged over 5 min. Data were downloaded weekly.

Weather Data Collection

An onsite weather station (model 6004-2, Unidata, Inc., Perth, Australia) was located on the prevailing upwind side of the feedyard. Data were collected for 2 m wind speed, wind direction, air temperature, solar radiation, and rainfall. A 10 m tower was located on the prevailing downwind side of the feedyard beginning in May 2007. The tower was instrumented at both 2 and 10 m with identical wind speed, wind direction, and ±0.1°C thermistors. Data were collected every 5 min and downloaded from both data loggers weekly. Selected data (wind speed, temperature, and rainfall) and mean deviations from long-term area means are summarized in table 1. Neither wind speed nor temperature were appreciably different from reported monthly means for the Amarillo area (NOAA, 2008) (table 1). A total of 435 mm of precipitation fell during the sampling period, which was slightly less than the average of 500 mm for this same period (table 1).

Data Quality Assurance/Quality Control

All data collected during each calibration cycle and 30 min after were removed from the data set. This allowed the analyzer to return to equilibrium. The TEI 17C operates best under room temperature (22°C) or cooler conditions. On two different occasions, the air cooling system malfunctioned during the warmest part of the summer, causing the temperature inside the lab trailer to exceed 32°C. This resulted in high temperatures in the convertor box of the TEI 17C, which could have affected the quality of the data; thus, those data were removed.

Emissions Modeling

Flux density was modeled using WindTrax ver. 2.0.7.8 (Thunder Beach Scientific, Nanaimo, British Columbia, Canada). WindTrax is a backward Lagrangian stochastic (bLs) model that predicts emissions based on random particle placement upwind of a concentration sensor. The bLs model

Table 1. Feedyard monthly mean and maximum wind speed and temperature and total precipitation.

Month	2 m Wind Speed ($m\ s^{-1}$) Mean	Deviation from Mean[a]	Max.	Ambient Temperature (°C) Mean	Deviation from Mean[a]	Max.	Precipitation (mm) Total	Deviation from Mean[a]
Mar. 2007	4.7	-0.8	15.3	10.5	1.7	28.9	88.2	59.5
Apr. 2007	5.8	0.3	20.5	11.5	-1.9	28.8	20.2	-13.6
May 2007	5.0	-0.2	16.8	18.2	-0.2	33.1	39.8	-23.7
June 2007	5.0	-0.1	22.2	22.7	-0.8	36.5	86.2	2.9
July 2007	3.8	-0.7	14.2	24.8	-1.9	36.2	59.4	-8.7
Aug. 2007	4.1	-0.2	12.2	25.1	0.5	41.3	14.0	-60.7
Sept. 2007	4.2	-0.4	13.3	21.6	1	36.4	96.0	48.2
Oct. 2007	4.7	0.1	17.1	15.7	1.1	32.9	1.2	-36.9
Nov. 2007	4.2	-0.5	16.0	7.5	0.2	27.6	0.6	-16.7
Dec. 2007	4.7	0.1	17.3	2.6	-0.2	24.0	18.6	3.1
Jan. 2008	5.8	1.2	20.0	1.9	-0.2	20.9	0.0	-16
Feb. 2008	5.1	0.1	19.6	5.4	0.6	24.2	11.2	-2.8
Annual average	4.8	-0.09		14.0	-0.01		435	-65

[a] Deviation from long-term means for Amarillo area (1892-2007) (NOAA, 2008).

simulates the transport of particles from a source to a measurement location, and predicts the ratio of the average concentration to the emission rate $(C/F)_{sim}$ (Flesch et al., 2004; Sommer et al., 2005). The emission rate is then inferred by the following relationship:

$$F_{bLs} = \frac{\overline{\chi_{obs}}}{\left(\frac{C}{F}\right)_{sim}} \tag{2}$$

where

$\overline{F_{bLs}}$ = inferred emission rate

$\overline{\chi_{obs}}$ = average ammonia concentration

$\left(\frac{C}{F}\right)_{sim}$ = ratio of concentration to emission rate.

WindTrax calculates the upwind trajectories of large numbers of particles based on wind and turbulence conditions. The ratio $(C/F)_{sim}$ is estimated by surface touchdowns of the particles:

$$\left(\frac{C}{F}\right)_{sim} = \frac{1}{N}\Sigma\left|\frac{2}{\omega_0}\right| \tag{3}$$

where N is the total number of computational particles released from the source area (the summation covers only touchdowns within the source area), and ω_0 is the vertical touchdown velocities.

As a minimum, WindTrax needs only four observations for calculations: mean wind speed, wind direction, roughness length, and atmospheric stability, which can be expressed in a number of ways (Gao et al., 2009). Flesch and Wilson (2005) conducted a sensitivity analysis of WindTrax and found it to be a robust model. They determined that some of its strengths included: experimental simplicity, absence of limitations on the size and shape of the source, and flexibility in the type and location of the concentration sampler. McBain and Desjardins (2005) found that WindTrax is relatively insensitive to minor upwind obstructions (e.g., fences, windbreaks, and water tanks if the distance between the obstruction and the concentration sensor is large). Sommer et al. (2005) evaluated WindTrax and found that this technique under-estimated NH_3 emissions from small plots by 16% to 24%. They attributed this error to the long averaging times (5 to 6 h) needed for sample collection. Several researchers have indicated that the bLs technique requires short sample integration times (15 to 60 min) so that the atmospheric stability requirements under the Monin-Obukhov similarity theory (MOST) are not violated (Harper et al., 2004, Flesch et al., 2007, 2004; Sommer et al., 2005; Todd et al., 2008).

Model input data consisted of 5 min averages of wind speed (at 2 and 10 m elevations above the pen surface), wind direction, ambient temperature, and NH_3 concentration. Pasquill-Gifford (P-G) stability class was also calculated on a 5 min time period. The surface roughness length (z_o) was set to 0.10 m following the study by Todd et al. (2008), who calculated it using sonic anemometer data from a 55,000-head capacity commercial feedyard in the Texas Panhandle. The U.S. Environmental Protection Agency (EPA) has also classified terrains in terms of effective z_o. Terrain with a z_o of 0.10 m is described as "low crop with occasional large obstacles, where the typical distance to the upwind obstacle divided by the height of the obstacle is >20 feet" (EPA, 2000).

WindTrax uses MOST, which applies to steady-state, horizontally homogenous conditions in the surface layer (EPA, 2000). The time scale for model inputs, therefore, should not exceed 1 h, nor should it be less than 5 min. This decreases the likelihood that assumptions under MOST will be violated. Temperature and wind speed measurements must be representative of a layer that is both high enough to be outside the influence of the surface roughness elements and low enough to be within the surface boundary layer. Typically, the measurements should be taken from 20 to 100 times z_o above the surface. Thus, for a z_o of 0.10 m, measurements should be taken at 2 to 10 m above the surface (EPA, 2000).

The pen source area was mapped into WindTrax with the aid of Google Earth (http://earth.google.com). Polygons of the pen were drawn over the satellite photograph. The pen source area was defined as fenced manure surfaces either occupied or recently occupied with cattle. All service roads and feed alleys were excluded from the source map. Also excluded from the source area were the runoff retention structure and the settling basins. Ammonia emissions from beef cattle runoff retention ponds have been determined to be <5% of total NH_3 emissions from a feedyard; thus, the retention pond was ignored as a source in this study (Flesch et al., 2007; Todd et al., 2008).

The stability class input was determined by use of the solar radiation/delta temperature (SRDT) method (EPA, 2000) (table 2). The vertical temperature gradient was determined by the differences between air temperature measurements at recommended heights of 2 and 10 m (EPA, 2000). The P-G stability classes range from very unstable (A) to neutral (D) to very stable (G). WindTrax tends to operate best when the stability is in the B to F range and does not work well under very unstable conditions, i.e., very sunny and low wind speeds. Figure 2 shows the frequency of stability classes that were used as inputs. Approximately 1% of the inputs into WindTrax occurred under very unstable conditions (fig. 2). These were not removed from the data set, as it was believed that they would have little impact on the overall mean estimated emission rates.

Table 2. Index of P-G stability class designations (EPA, 2000).

Daytime				
Wind Speed ($m\ s^{-1}$)	Solar Radiation ($W\ m^2$)			
	>925	925-675	675-175	<175
<2	A	A	B	D
2 to 3	A	B	C	D
3 to 5	B	B	C	D
5 to 6	C	C	D	D
>6	C	D	D	D

Nighttime		
Wind Speed ($m\ s^{-1}$)	Vertical Temperature Gradient (10 m temp. - 2 m temp.)	
	<0	>0
<2.0	E	F
2.0 to 2.5	D	E
>2.5	D	D

Table 3. Summary of monthly NH_3 concentrations (ppm).

Month	N	Minimum	Lower Quartile	Median	Mean	Upper Quartile	Maximum
Mar. 2007	8160	0.006	0.332	0.435	0.498	0.636	1.904
Apr. 2007	7682	0.050	0.334	0.533	0.594	0.822	1.835
May 2007	6077	0.024	0.434	0.574	0.568	0.732	1.460
June 2007	8164	0.001	0.236	0.464	0.684	0.815	5.108
July 2007	8123	0.008	0.465	0.642	0.702	0.873	2.891
Aug. 2007	2455	0.094	0.414	0.604	0.773	1.037	3.444
Sept. 2007	8162	0.018	0.460	0.672	0.755	0.974	5.448
Oct. 2007	8726	0.083	0.317	0.480	0.527	0.690	1.940
Nov. 2007	3020	0.004	0.264	0.413	0.453	0.563	1.973
Dec. 2007	8880	0.003	0.114	0.274	0.373	0.494	2.395
Jan. 2008	6490	0.103	0.254	0.402	0.509	0.689	1.544
Feb. 2008	7698	0.053	0.253	0.363	0.439	0.574	1.294

Results and Discussion

Ambient Concentrations

Monthly NH_3 minimum, quartile, median, mean, and maximum concentrations are presented in table 3. Ammonia concentrations tended to be greatest in the summer months (June, July, August) and lowest in the winter months (December, January, February). Ammonia concentrations in the spring (March, April, and May) and autumn (September, October, and November) were about the average of the summer and winter months, and were very close to the annual average.

Individual 5 min NH_3 concentrations exceeded 2.0 ppm on 22 days. Daily averages exceeded 1.0 ppm on 18 days, and monthly averages never exceeded 0.8 ppm. There were five days in June 2007 (days of the year 172 to 176), ten days in August 2007 (days of the year 214 to 219 and 222 to 225), and three days in September 2007 (days of the year 255, 264, and 267) during which daily averages exceeded 1.0 ppm. Wind speeds for those 18 days ranged from 2.6 to 5.0 m s^{-1}, which was at, or below, the mean wind speed at the feedyard (table 1). Ambient temperatures for those 18 days ranged from 19.5°C to 26.1°C, which were also near the average at the site (table 1). A considerable amount of manure was harvested from the pen between July and September 2007. It was probable that disturbance of the manure pack due to manure harvesting, coupled with low wind speeds that occurred upwind of the sampler, contributed to higher NH_3 concentrations.

Typical rural NH_3 concentrations have been reported to be approximately 0.7 ppb (WHO, 1986). In contrast, Auvermann (2008, unpublished data) reported background NH_3 concentrations in the Texas Panhandle as an annualized average of 0.0045 ppm.

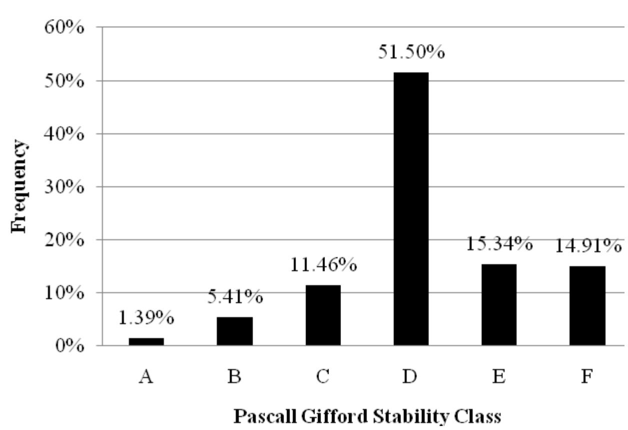

Figure 2. Histogram showing the P-G stability classes used in the emissions estimates.

Diel Concentration Variation

Diel variation (i.e., variation occurring over a 24 h period) in NH_3 concentrations was evident across all months. Concentrations were least during early morning hours and greatest during early afternoon across all seasons (fig. 3). Wintertime low concentrations were lower than all other seasonal low concentrations by about one-half. All seasons had similar high concentrations of around 0.8 ppm. A majority of the manure harvesting activity took place during the warmer periods of the spring and summer. It is probable that disturbance of the manure surface from the harvesting equipment caused elevated NH_3 concentrations, as there

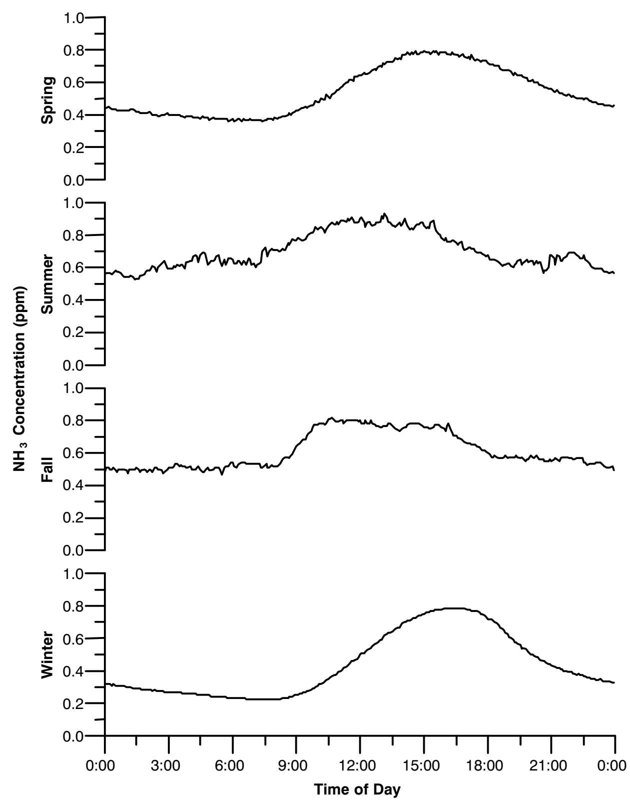

Figure 3. Mean 5 min diel NH_3 concentrations (ppm) for the spring (March, April, May), summer (June, July, August), fall (September, October, November), and winter (December, January, February) months.

Table 4. Summary of monthly NH_3 flux densities ($\mu g\ m^{-2}\ s^{-1}$).

Month	*N*	Minimum	Lower Quartile	Median	Mean	Upper Quartile	Maximum
Mar. 2007	8160	0.7	35.1	51.2	57.8	75.3	271.0
Apr. 2007	7682	0.5	35.7	72.0	108.9	134.5	711.3
May 2007	6077	0.3	28.7	61.8	86.3	97.2	324.4
June 2007	8164	0.1	27.5	54.4	85.3	107.5	845.7
July 2007	8123	0.3	34.9	60.8	74.6	101.1	409.7
Aug. 2007[a]	2456	0.8	18.6	40.4	71.6	86.6	678.0
Sept. 2007	8162	0.3	36.9	64.0	89.2	115.2	667.4
Oct. 2007	8726	2.2	27.1	46.1	57.4	74.9	277.5
Nov. 2007[a]	3020	0.5	20.3	33.9	42.7	55.8	293.5
Dec. 2007	8880	0.2	11.1	28.3	45.0	58.6	415.7
Jan. 2008	6490	1.9	28.3	50.0	74.1	97.4	363.5
Feb. 2008	7698	1.7	21.7	42.4	55.6	71.1	274.4
Mean					70.7		

[a] Partial data loss due to equipment failure.

were several NH_3 spikes during this time period. However, no documentation of manure harvesting in relation to the concentration sensor was documented.

Flux Density

Monthly NH_3 minimum, quartile, median, mean, and maximum flux densities are presented in table 4. Seasonal variation of NH_3 flux density followed the same pattern as NH_3 concentrations. Summer fluxes were greater than winter, with autumn fluxes being intermediate. Other researchers have reported that cold air temperatures decrease NH_3 emissions (Adriano et al., 1974; Todd et al., 2008; Cole et al., 2009). Mean monthly ammonia flux density ranged from a low of 42.7 $\mu g\ m^{-2}\ s^{-1}$ to a high of 108.9 $\mu g\ m^{-2}\ s^{-1}$ in November and April 2007, respectively. The large flux densities observed in the spring months were not anticipated. Annual mean flux density was 70.7 $\mu g\ m^{-2}\ s^{-1}$ (table 4). This is consistent with Todd et al. (2008), who reported a summertime emission rate of 70 $\mu g\ m^{-2}\ s^{-1}$ for a Texas feedyard and somewhat lower than McGinn et al. (2007), who reported an average annual emission rates of 84 $\mu g\ m^{-2}\ s^{-1}$ for a similarly sized feedyard in Canada.

April flux densities were substantially greater than all other months (table 4), although ambient concentrations were not greater (table 4). April was a very dry, windy month, receiving only 20 mm of precipitation as compared with the historical average precipitation of 33.8 mm. Air temperature ranged from -3.3°C to 28.9°C, with an overall monthly mean of 11.4°C. Wind speed more than likely offered the greatest influence to elevated fluxes. Winds ranged from 0 to 74 km h^{-1} with a mean of 20.7 km h^{-1} (5.8 m s^{-1}). There were 15 days with maximum wind speeds greater than 40 km h^{-1} (11.1 m s^{-1}). In addition, precipitation in March was approximately three times the normal rainfall. It is probable that NH_3 was held in the aqueous phase in March, resulting in a "flush" of NH_3 emissions in April.

Todd et al. (2008) conducted six field campaigns beginning in the summer of 2002 and ending in the spring of 2005 at a 55,000-head Texas feedyard. They measured NH_3 concentrations using flux gradient methods through three summers, two winters, and one spring season. They found that winter flux densities were about 50% of summer flux densities. In contrast, we found winter emission rates to be about 75% of summer rates based on one calendar year.

Diel Flux Density Variation

Diel variation (i.e., variation occurring over a 24 h period) in NH_3 flux densities was evident across all months. Flux densities were least during early morning hours and greatest during early afternoon across all seasons (fig. 4). Winter time low concentrations were lower than all other seasonal low concentrations by about one-half. Spring, summer, and fall flux densities all peaked at about 140 $\mu g\ m^2\ s^{-1}$, while winter flux density peaked around 120 $\mu g\ m^2\ s^{-1}$ (fig. 4).

McGinn et al (2007) saw similar diel patterns from a feedyard in Canada, although they had a much larger range of flux density than what we found. That would most likely

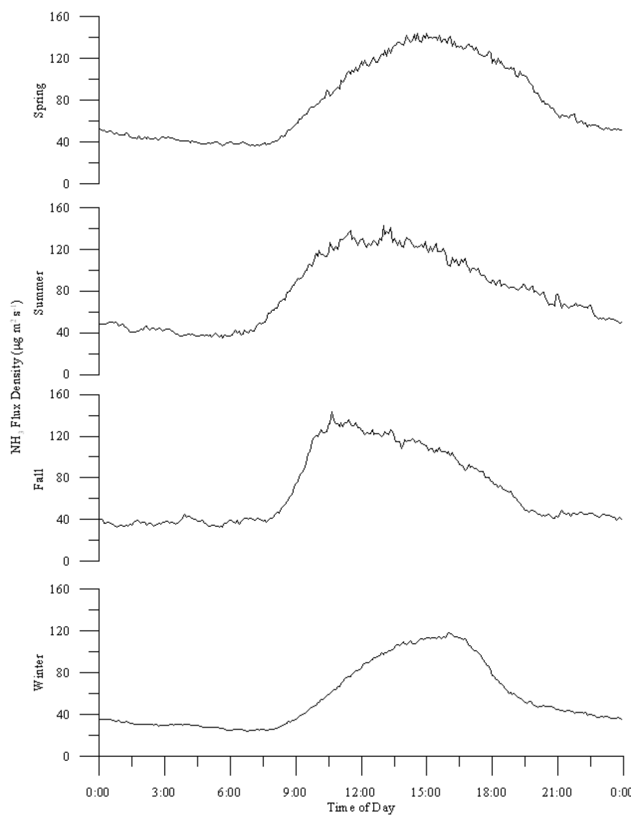

Figure 4. Mean 5 min diel NH_3 flux density ($\mu g\ m^2\ s^{-1}$) for the spring (March, April, May), summer (June, July, August), fall (September, October, November), and winter (December, January, February) months.

Table 5. Nitrogen balance showing NH_3-N volatilization as percentage of fed nitrogen.

Month	No of. Head on Feed	NH_3-N Emission Rate (kg d^{-1})	Fed N (kg d^{-1})	NH_3-N Volatilization as % of Fed N
Mar. 2007	18,581	1387	3182	43.6
Apr. 2007	18,787	2613	3074	85.0
May 2007	19,880	2073	3222	64.3
June 2007	19,373	2047	3718	55.1
July 2007	19,717	1802	3837	47.0
Aug. 2007	19,316	1718	3699	46.4
Sept. 2007	19,026	2141	3606	59.4
Oct. 2007	21,214	1378	3533	39.0
Nov. 2007	20,098	1025	3568	28.7
Dec. 2007	21,256	1079	3446[a]	31.3
Jan. 2008	21,627	1778	3446[a]	51.6
Feb. 2008	20,136	1334	3446[a]	38.7
Annual mean	19,918	1,698	3,481	48.8

[a] Estimated fed N as average of other months.

be attributable to differences in environmental conditions between feedyard locations.

Nitrogen Mass Balance

Modeled NH_3 emissions from the pen surface were compared with the mass balance of N for the feedyard. Monthly ration samples were collected from feedbunks and analyzed for total N. Total fed N was calculated based on diet N concentration and the total feed fed in the feedyard on a monthly basis. The NH_3-N daily emission rate was calculated from the modeled flux densities and compared with the average daily fed N rate. The total fraction of fed N lost as NH_3-N averaged 48.8% (table 5), with monthly values ranging from 29% to 64% for all months except April, which was 85.0%. However, March 2007 was a wetter month than normal (table 1), so it was hypothesized that the wetter pen surface actually held the NH_3 in an aqueous phase. Then, as the pen surface dried, a "flush" of NH_3-N was emitted in April. This agrees with other work that suggests that NH_3 moves with moisture (Ernst and Massey, 1960; Cole et al., 2009).

The fraction of N lost as NH_3 (48.8%) in this research was similar to the 45% reported by Todd et al. (2008), lower than the 63% to 65% loss reported by Flesch et al. (2007) for Texas feedyards, and lower than the 63% reported by McGinn et al. (2007) for a feedyard in Canada. Erickson et al. (1999) reported 52% to 63% of fed N was lost as NH_3 via volatilization at an experimental feedlot in Nebraska, which compares well with what we observed.

NH_3 Pen Surface Emission Coefficient

Emission factors (EFs) are used to by regulatory agencies to determine pollutant discharges. Beginning 20 January 2009, animal agricultural operations emitting 100 lbs or more per day of either NH_3 or hydrogen sulfide (H_2S) were required to report these emissions from their facilities under the Emergency Planning and Community Right-to-Know Act (EPCRA). Under this ruling, a beef cattle feeding operation in Texas would begin reporting emissions when they reached 625 animals. This feedyard falls well within that criterion. The EPA lists NH_3 as a compound of concern, as it has been determined that NH_3 in the presence of oxides of nitrogen and sulfur (NO_x and SO_x, respectively) is a precursor for the production of fine particulate matter (PM). Therefore, accurate emission factors that reflect both climate and management practices need to be in place.

Determination of accurate whole-farm EFs can be a difficult, time consuming, and expensive prospect. Several approaches are listed in the literature, ranging from mathematical to labor-intensive micrometeorological techniques. Among the simplest is that listed by the European Environmental Agency (EEA) (Van Der Hoek, 1998) for beef animals in Europe. The EEA determined EFs by multiplying average N excretion per animal by an NH_3 volatilization percentage for NH_3 loss for housing, manure storage, manure spreading, and grazing. The summation of these four categories resulted in total emissions estimates of 14.3 kg NH_3 animal^{-1} year^{-1} for beef cattle. These factors were calculated for what was determined to be one "average" animal present on site for 365 days. The EEA also determined that uncertainties in ammonia emission factors were in the range of 30%.

Emission factors should account not only for the species of animal but also the management (types of feed, stocking density, pen drainage, manure harvesting schedule), production stage (cow-calf vs. stocker vs. finishing), and environmental conditions and climate (semi-arid vs. tropical, rainfall, wind speed). Thus, emission factors need to be geographically as well as species specific, while accounting for management practices.

The development of an EF requires measurements over a variety of conditions, such as summer vs. winter, or wet vs. dry conditions. As is evident in table 1, climate conditions can vary greatly, even on a monthly basis. However, when averaged over an annual basis, deviations may not be as great. Climate unquestionably determines NH_3 emissions from feedyards. Several researchers have shown that summer emissions tend to be about two times the winter emissions (Erickson et al., 1999; Flesch et al., 2007; Todd et al., 2008). From this, it is evident that short-term measurements or measurements taken during a single season will more than likely be inadequate for determining an effective and useful EF. The data presented in table 1 indicate that, although monthly averages can and did deviate from normal, yearly averages were very close to normal. This would indicate that the study period should be representative of climate conditions for the geographic area.

In the present study, a pen surface emission coefficient (EC) is presented rather than a whole-farm emission factor in view of the fact that the retention pond, settling basins, and potential losses from land application are not accounted for. The EC was calculated based on the number of head fed in one year by the feedyard. The estimated annualized mean daily emission rate of 1,698 kg d^{-1} was divided by the annualized mean head on feed (table 5). This resulted in an emission coefficient of 85.3 g head^{-1} d^{-1} from the pen surface (table 6).

For comparison, NH_3-N EFs reported by other researchers are presented in table 6. Although there are several listed that were derived in Europe, caution must be utilized in making these comparisons. The management of cattle in Europe differs considerably from management in a typical U.S. beef cattle feedyard. Normal European management practices incorporate both a "stable" and a "pasture" combination for cattle. Cattle are kept in a stable about 40% of the year and are on pasture the remaining 60%. During the stable rotation,

Table 6. Comparison of NH_3-N emission factors.

Source	NH_3-N Emission Factors (g head^{-1} d^{-1})	Method	Calculation Basis	Location
Battye et al. (1994)	18.5 (annual)	Literature	Production	Europe
Asman (1992)	18.9 (annual)	Literature	Production	Europe
Hutchinson et al. (1982)	32.8 (spring/summer)	Micrometeorological	Capacity	Colorado
Buijsman et al. (1987)	33.4 (annual)	Literature	Production	Europe
Todd et al. (2008)	44 (annual)	Micrometeorological	Capacity	Texas
Rhoades et al. (this study)	85.3 (pens only; annual)	Inverse dispersion	Production	Texas
McGinn et al. (2007)	114.8 (summer/fall)	Micrometeorological / Inverse dispersion	Capacity	Canada
Flesch et al. (2007)	123 (summer only)	Micrometeorological	Production	Texas

cattle are fed a supplementary diet of about 13% protein, which is very similar to what is fed in a typical U.S. beef cattle feedlot, while during the pasture rotation, cattle are grazing, often on very high quality, high protein forages, with little or no dietary supplementation.

It is apparent from table 6 that, depending on the source, a wide range of emission factors is found in the literature. Emission factors can vary greatly depending on climate, management, and measurement methodologies. The EPA (2005) based its emission factors on two studies comprised of limited data for drylot steers and heifers.

A lack of consistency in the calculation basis of an EF is apparent from table 6. This lack of standardization can lead to difficulty in making direct comparisons of results. A capacity-based EF is calculated on the permitted number of head that a facility is allowed multiplied by the number of "turnovers" that is typical of that facility. For example, a feedyard with a permitted capacity of 25,000 head that has an average feeding time of 150 days will be able to feed a total capacity of 60,833 head year^{-1}. This can typically lead to a lower emission rate on a per-head basis, as most feedyards are unable to operate at full capacity. Another method is to base the calculation on a production scale. This requires a good working relationship with the feedyard, as the actual number of animals on feed is required. This will typically lead to a higher emission rate on a per-head basis. To this point, little work has been to determine which method will yield the most accurate and useful result.

Todd et al. (2008) differentiated source area from a feedyard on the Southern High Plains. The feedyard studied by Todd et al. (2008) had approximately two times the feeding capacity of the feedyard studied here, although similar types of cattle were fed in both yards. Todd et al. (2008) based their EF calculations on a capacity basis, i.e., total one-time capacity (45,000 head) × turnovers (2.25 per year). Conversely, we calculated a pen EC on a production basis (actual number of animals fed in a year). By recalculating Todd's numbers to a production basis (i.e., 4430 kg NH_3-N d^{-1} / 44,651 head on the yard at time of sampling), an EF of 99.2 g NH_3-N head^{-1} d^{-1} results, which compares well with the EC determined in this study.

Caution is urged in the application of this EF to circumstances that may vary significantly from conditions found in this area. The feedyard used in this study was typical of the feedyards in the area in management of cattle, manure, and diet formulation. The EF presented here should be a good indication of NH_3 emissions when averaged over a year for similarly managed feedyards. The introduction of other feedstuffs, such as wet distillers grains, into the diet can potentially effect NH_3 emissions, especially if the diets have an increase in CP.

Conclusions

The following conclusions were drawn from this research:

- Ambient NH_3 concentrations measured at 3.3 m height in the center of a feedyard for a calendar year ranged from 0.37 to 0.77 ppm, with an average annual concentration of 0.57 ppm.
- Flux densities calculated using a backwards Lagrangian stochastic model ranged from 42.7 to 123.1 μg m^2 s^{-1}, with a mean of 70.7 μg m^{-2} s^{-1}.
- A per-head pen surface emission coefficient was calculated as 85.3 g NH_3-N (head fed)$^{-1}$ day^{-1}.

Additional research questions were raised during this study. Several methods are available for NH_3 measurement; however, at this time they are not comparable, as different methods can provide different results (Harper et al., 2004; Hudson and Ayoko, 2008). Greater understanding of comparative measurement techniques is critical to aid both producers and regulatory agencies. While several researchers have established that overfeeding of N can result in greater production of NH_3, little work has been done to establish best management practices that meet the nutritional requirement of the animals, reduce negative environmental and social impacts, and remain economically feasible. Development of a process-based model that accurately describes movement of N through a feedyard setting would be a useful tool for describing areas that could be inefficient in any of the above areas.

Acknowledgements

The authors wish to acknowledge the managers of the participating feedlot for providing space to locate sampling equipment and for supplying supplementary data. This project was funded by USDA-CSREES Project No. TS-2006-06009 entitled "Air Quality: Odor, Dust, and Gaseous Emissions from Concentrated Animal Feeding Operations in the Southern Great Plains."

References

Adriano, D. C., A. C. Chang, and R. Sharpless. 1974. Nitrogen loss from manure as influenced by moisture and temperature. *J. Environ. Qual.* 3(3): 258-261.

Asman, W. A. H. 1992. Ammonia emission in Europe: Updated emission and emission variations. Bilthoven, The Netherlands: National Institute of Public Health and Environmental Protection. Available at: www.rivm.nl/bibliotheek/rapporten/228471008.html. Accessed 15 January 2009.

Ball, R., D. R. Keeney, P. W. Theobald, and P. Nes. 1979. Nitrogen balance in urine-affected areas of a New Zealand pasture. *Agron. J.* 71(2): 309-314.

Battye, R., W. Battye, C. Overcash, and S. Fudge. 1994. Development and selection of ammonia emission factors. Final report, February-August 1994. EPA Contract Number 68-D3-0034. Research Triangle Park, N.C.: U.S. EPA Atmospheric Research and Exposure Assessment Laboratory.

Buijsman, E., H. F. M. Maas, and W. A. H. Asman. 1987. Anthropogenic NH_3 emissions in Europe. *Atmos. Environ.* 21(5): 1009-1022.

Cole, N. A., R. N. Clark, R. W. Todd, C. R. Richardson, A. Gueye, L. W. Greene, and K. McBride. 2005. Influence of dietary crude protein concentration and source on potential ammonia emissions from beef cattle manure. *J. Animal Sci.* 83(3): 722-731.

Cole, N. A., A. M. Mason, R. W. Todd, M. B. Rhoades, and D. B. Parker. 2009. Chemical composition of pen surface layers of beef cattle feedlots. *Prof. Animal Scientist* 25(5): 541-552.

Duyson, R., G. Erickson, D. Schulte, and R. Stowell. 2003. Ammonia, hydrogen sulfide, and odor emissions from a beef cattle feedlot. ASAE Paper No. 034109. St. Joseph, Mich.: ASAE.

EPA. 2000. Meteorological monitoring guidance for regulatory modeling applications. EPA Report No. EPA-454/R-99-005. Research Triangle Park, N.C.: U.S. EPA Office of Air Quality Planning and Standards.

EPA. 2005. National emissions inventory - Ammonia emissions from animal agricultural operations. Revised draft report. Washington, D.C.: U.S. EPA. Available at: www.epa.gov/ttnchiel/ap42/ch09/related/nh3inventorydraft_jan2004.pdf. Accessed 12 January 2009.

Erickson, G., T. Klopfenstein, T. Milton, and D. Herold. 1999. Effects of matching protein to requirements on performance and waste management in the feedlot. 1999 Nebraska Beef Cattle Report. Lincoln, Neb.: University of Nebraska. Available at: http://digitalcommons.unl.edu/animalscinbcr/402/. Accessed 11 April 2008.

Ernst, J. W., and H. F. Massey. 1960. The effects of several factors on volatilization of ammonia formed from urea in the soil. *SSSA J.* 24(2): 87-90.

Flesch, T. K., and J. D. Wilson. 2005. Chapter 22: Estimating tracer emissions with a backward Lagrangian stochastic method. In *Micrometerology in Agricultural Systems*, 512-531. M. K. Viney, ed. Madison, Wisc.: ASA-CSSA-SSSA.

Flesch, T. K., J. D. Wilson, L. A. Harper, B. P. Crenna, and R. R. Sharpe. 2004. Deducing ground-to-air emissions from observed trace gas concentrations: A field trial. *J. Applied Meteorol.* 43(3): 487-503.

Flesch, T. K., J. D. Wilson, L. A. Harper, R. W. Todd, and N. A. Cole. 2007. Determining ammonia emissions from a cattle feedlot with an inverse dispersion technique. *Agric. Forest Meteorol.* 144(1-2): 139-155.

Frank, B., and C. Swensson. 2002. Relationship between content of crude protein in rations for dairy cows and milk yield, concentration of urea in milk, and ammonia emissions. *J. Dairy Sci.* 85(7): 1829-1838.

Gao, Z., M. Mauder, R. L. Desjardins, T. K. Flesch, and R. P. van Haarlem. 2009. Assessment of the backward Lagrangian stochastic technique for continuous measurements of CH_4 emissions. *Agric. Forest Meteorol.* 149(9): 1516-1523.

Harper, L. A., N. A. Cole, R. Todd, T. K. Flesch, and R. R. Sharpe. 2004. Ammonia emissions from beef feeding operations: Measurement technologies and emissions. In *Proc. Plains Nutrition Council*, 27-50. Publ. No. AREC 04-14. Amarillo, Tex.: Texas A&M Research and Extension Center.

Hausinger, R. P. 2004. Metabolic versatility of prokaryotes for urea decomposition. *J. Bacteriology* 186(9): 2520-2522.

Hudson, N., and G. A. Ayoko. 2008. Odour sampling: 2. Comparison of physical and aerodynamic characteristics of sampling devices: A review. *Bioresource Tech.* 99(10): 3993-4007.

Hutchinson, G. L., A. R. Mosier, and C. E. Andre. 1982. Ammonia and amine emissions from a large cattle feedlot. *J. Environ. Qual.* 11(2): 288-293.

Koziel, J. A., D. B. Parker, B.-H. Baek, K. J. Bush, M. Rhoades, and Z. Perschbacher-B. 2005. Ammonia and hydrogen sulfide flux from beef cattle pens: Implication for air quality measurement methodologies and evaluation of emission controls. In *Livestock Environment VII: Proc. 7th Intl. Symposium*, 402-410. St. Joseph, Mich.: ASAE.

McBain, M. C., and R. L. Desjardins. 2005. The evaluation of a backward Lagrangian stochastic (bLs) model to estimate greenhouse gas emissions from agricultural sources using a synthetic tracer source. *Agric. Forest Meteorol.* 135: 61-72.

McGinn, S. M., T. K. Flesch, B. P. Crenna, K. A. Beauchemin, and T. Coates. 2007. Quantifying ammonia emissions from a cattle feedlot using a dispersion model. *J. Environ. Qual.* 36(6): 1585-1590.

NOAA. 2008. Climate notes for the Amarillo area. Amarillo, Tex.: National Weather Service Weather Forecast Office. Available at: www.srh.noaa.gov/ama/?n=rec_norm_ama. Accessed 12 May 2008.

Sommer, S. G., S. M. Mcginn, and T. K. Flesch. 2005. Simple use of the backwards Lagrangian stochastic dispersion technique for measuring ammonia emission from small field-plots. *European J. Agron.* 23(1): 1-7.

Stewart, B. A. 1970. Volatilization and nitrification of nitrogen from urine under simulated cattle feedlot conditions. *Environ. Sci. and Tech.* 4(7): 579-582.

TCFA. 2008. The impact of cattle feeding on the TCFA area economy. Amarillo, Tex.: Texas Cattle Feeders Association. Available at: www.tcfa.org/impact.html. Accessed 16 May 2008.

Todd, R. W., N. A. Cole, and R. N. Clark. 2006. Reducing crude protein in beef cattle diets reduces ammonia emissions from artificial feedlot surfaces. *J. Environ Qual.* 35(2): 404-411.

Todd, R. W., N. A. Cole, R. N. Clark, T. K. Flesch, L. A. Harper, and B.-H. Baek. 2008. Ammonia emissions from a beef cattle feedyard on the southern High Plains. *Atmos. Environ.* 42(28): 6797-6805

Vallis, I., L. A. Harper, V. R. Catchpoole, and K. L. Weier. 1982. Volatilization of ammonia from urine patches in a subtropical pasture. *Australian J. Agric. Res.* 33(1): 97-107.

Van Der Hoek, K. W. 1998. Estimating ammonia emission factors in Europe: Summary of the work of the UNECE ammonia expert panel. *Atmos. Environ.* 32(3): 315-316.

Vasconcelos, J. T., and M. L. Galyean. 2007. Nutritional recommendations of feedlot consulting nutritionists: The 2007 Texas Tech University survey. *J. Animal Sci.* 85(10): 2772-2781.

Whitehead, D. C., and N. Raistrick. 1991. Effects of some environmental factors on ammonia volatilization from simulated livestock urine applied to soil. *Biol. Fert. Soils* 11(4): 279-284.

WHO. 1986. Air quality guidelines for Europe. WHO Regional Publications, European Series. Geneva, Switzerland: World Health Organization. Available at: www.who.int/phe/health_topics/outdoorair_aqg/en/.