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Ammonia Emission from a Beef Cattle Feedlot and Its Local Dry Deposition and Re-Emission

S. M. McGinn,* H. H. Janzen, T. W. Coates, K. A. Beauchemin, and T. K. Flesch

Abstract

Ammonia (NH_3) volatized from livestock manure is affiliated with ecosystem and human health concerns and decreased fertilizer value of manure and can also be an indirect source of greenhouse gas. Beef cattle feedlots, where thousands of cattle are grouped together to enable greater control of feed management and production, are hot spots in the agricultural landscape for NH_3 emissions. Quantifying the feedlot NH_3 emissions is a difficult task, partly due to the reactive nature of NH_3 within and surrounding the feedlot. Our study used a dispersion model coupled to field measurements to derive NH_3 emissions from a feedlot in southern Alberta, Canada. The average feedlot NH_3 emission was 50 μg m^{-2} s^{-1} (85 g $animal^{-1}$ d^{-1}), which coincides with a low dietary crude protein content. At a location 165 m east of the feedlot, a flux gradient (FG) technique measured an average NH_3 deposition of 12.0 μg m^{-2} s^{-1} (west wind) and 5.3 μg m^{-2} s^{-1} (east wind). Ammonia FG emission averaged 1 μg m^{-2} s^{-1} with east winds, whereas no NH_3 emission was found for west wind. Using soil-captured NH_3, there was a decrease in deposition with distance from the feedlot (50% over 200 m). Collectively, the results of this study provide insight into the dynamics of NH_3 in the agricultural landscape and illustrate the need for NH_3 mitigation to improve the environmental and economic sustainability of cattle feedlots.

Core Ideas

- Beef feedlots are "hot spots" of ammonia emissions in the landscape.
- A significant fraction of the emitted ammonia is deposited to local land.
- Nitrogen fate of emitted ammonia is dynamic.

Agriculture is a major source of atmospheric ammonia (NH_3). Globally, it is estimated that total NH_3 nitrogen (N) emission is 54 Tg yr^{-1}, of which domestic animal manure accounts for 40% and synthetic N fertilizers contribute 17% (Bouwman et al., 1997). In Canada, livestock manure and synthetic fertilizer account for 90% of the 495 Gg of NH_3 emitted annually (Environment Canada, 2014). In western Canada, intensive beef cattle feedlots, which accumulate cattle manure, are "hot spots" of NH_3 emissions on the agricultural landscape.

The retention of dietary N by cattle is low. As a result, about 214 g N are excreted daily per feedlot animal (based on 500 kg body weight) (Arogo et al., 2006). Much of the N is excreted as urinary urea, which is rapidly converted to NH_3 by urease (Voorburg and Kroodsma, 1992). In open cattle feedlots, upward of 60% of the fed N may be lost to the atmosphere as NH_3 (Flesch et al., 2007). It follows that a 22,500 animal feedlot emits about 2.5 Mg of NH_3–N d^{-1} (McGinn et al., 2007), with a daily economic value of over $4,000 (CAN$) if it were to be replaced with urea fertilizer [cost of N emission calculated as (2590 kg NH_3–N d^{-1}) × ($750 Mg^{-1} urea) × (Mg urea 460 kg^{-1} N)].

Practices that decrease NH_3 emissions from manure would improve air quality, especially in confined air sheds and locations adjacent to intensive livestock facilities. On a local scale, Loubet et al. (2006) estimated that between 2 and 60% of NH_3 emitted is deposited within 1 km of the source. Dry deposition occurs when the surface NH_3 concentration is less than that in the air (Cape, 2014). However, when this NH_3 concentration gradient reverses, NH_3 is emitted from the surface back to the air. This bidirectional transfer of NH_3 is more dynamic when the surface is wet. Although NH_3 has a high affinity for water, when water evaporates, the dissolved NH_3 may be re-emitted via the gas phase as the water evaporates at the surface (Cape, 2014).

Local dry deposition of NH_3, which diminishes with increasing distance from the source, may benefit crops in N-deficient soil or can lead to excessive N accumulation in soils, leading to acidification of soil (Sanderson et al., 2006) and changes in species diversity of natural ecosystems (Sutton et al., 1993).

In the atmosphere, NH_3 is a precursor of fine particulate matter, largely in the form of particulate ammonium (NH_4^+),

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*Corresponding author (sean.mcginn@agr.gc.ca).

S.M. McGinn, H.H. Janzen, and K.A. Beauchemin, Agriculture and Agri-Food Canada, Lethbridge, AB, Canada T1J 4B1; T.W. Coates, Univ. of Melbourne, Melbourne, VIC, Australia 3010; T.K. Flesch, Univ. of Alberta, Edmonton, AB, Canada. Assigned to Associate Editor April Leytem.

Abbreviations: CP, crude protein; DM, dry matter; DMI, dry matter intake; DOY, day of year; DV, deposition velocity; FG, flux gradient; OPL, open-path laser; SC, soil-capture.

which is involved in aerosol formation. These aerosols contribute to poor air quality (Erisman and Schaap, 2004; McCubbin et al., 2002) especially in confined air sheds, causing visibility degradation (Barthelmie and Pryor, 1998). The airborne particulate matter also poses a health risk to people (Popendorf et al., 1985) and cattle (MacVean et al., 1986). Airborne NH_x (NH_3 + NH_4^+) can be transported long (continental-scale) distances and is therefore a transboundary pollutant (Ferm, 1998; Park et al., 2004). At this scale, Asman and van Jaarsveld (1992) indicated in a modeling approach that about half of the emitted NH_3 is deposited as NH_3 and the remaining half is deposited as NH_4^+.

The objectives of our study were to measure (i) NH_3 emission from a beef cattle feedlot and its percentage of N ingested by cattle, (ii) local dry deposition of NH_3 to land downwind of the cattle feedlot, and (iii) local re-emission of NH_3 from land adjacent to a cattle feedlot. These three objectives are expected to clarify the fate of NH_3–N from beef cattle manure at feedlots.

Materials and Methods

Site Description

The study was initiated on 20 June 2014 (day of year [DOY] 171) and ended on 6 Nov. 2014 (DOY 310); however, between 14 Aug. 2014 (DOY 226) and 1 Oct. 2014 (DOY 274) no measurements were made due to harvest and seeding activities. The study was conducted at a commercial beef cattle feedlot near Lethbridge, Alberta (49°45′ N, 112°38′ W; elevation, 863 m). The land to the east of the feedlot was initially planted to canola (*Brassica rapa* L.) and later (in September) planted to winter wheat (*Triticum aestivum* L.). A pasture was located to the west of the feedlot (Fig. 1) where no detectable source of NH_3 was measured (i.e., no elevated NH_3 concentration with west winds) (Fig. 2). The topography was flat, with few obstructions that could alter the wind flow, except for 3-m-high solid fencing between pens used to shelter cattle during winter. Sprinklers were used periodically in July to reduce pen dust. The predominant wind direction was southwest in summer and more westerly in the fall. During the measurement campaign, the average air temperature (± SD) was 15 ± 7°C, relative humidity was 65 ± 19%, and the average wind speed was 3 ± 2 m s^{-1}; the maximum 10-min gust was 14 m s^{-1}. The average live weight of the cattle was 504 ± 22 kg, and the number of cattle in the feedlot was 8244 ± 371. The total pen area of the feedlot (all pens were occupied) was 162,600 m^2, resulting in an average area per animal of 20 m^2. The manure in each pen was frequently piled in the center of each pen. The manure was periodically removed and piled in the northeast section of the feedlot (Fig. 1). This potential source of NH_3 was excluded by filtering the wind direction when calculating feedlot emission.

Cattle Diet Composition

Several diets were fed simultaneously to cattle (Table 1). The cattle entering the feedlot were transitioned from a high-forage diet of 54% corn silage (dry matter [DM] basis; Diet 1) to a high-grain diet consisting of 85% barley (DM basis; Diet 5). A few weeks before slaughter, the cattle were fed two diets that promoted weight gain (Diets 6 and 7). Later in the study, the diets (Table 1) included dry distillers' grain with less barley grain. The percent crude protein (CP; DM basis) of each diet was calculated using National Research Council (2000) values. The weekly averaged number of cattle on each diet and the dry matter intake (DMI) were determined from the daily values as recorded by a feedlot management system. The N intake of the cattle for each week and each diet (Table 2) was derived from the CP, number of cattle, and DMI [kg N = (kg DMI) × (kg CP/kg DMI) × (kg N/6.25 kg CP)].

Feedlot-based Ammonia Emission

Two open-path lasers (OPLs) for NH_3 (GasFinder2.0, Boreal Lasers) were initially set up side by side on the east side of a 500-animal research feedlot (Agriculture and Agri-Food Canada's Lethbridge Research Centre, Alberta) to correct for the between-sensor differences. Over a 3-d period, the continuous 10-min averaged NH_3 mixing ratios (ppm_v) were recorded over a path length of 100 m. The OPL readings varied between 0 and 0.3 ppm_v after each OPL was corrected for air pressure and

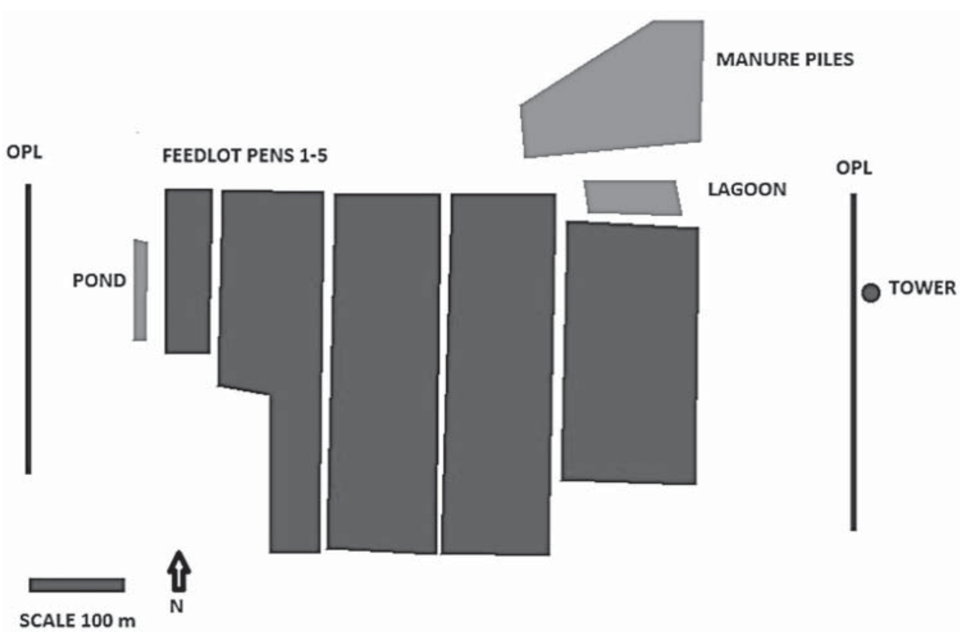

Fig. 1. Study site showing open-path laser (OPL) locations and position of the concentration profile (dot) colocated with a three-dimensional sonic anemometer.

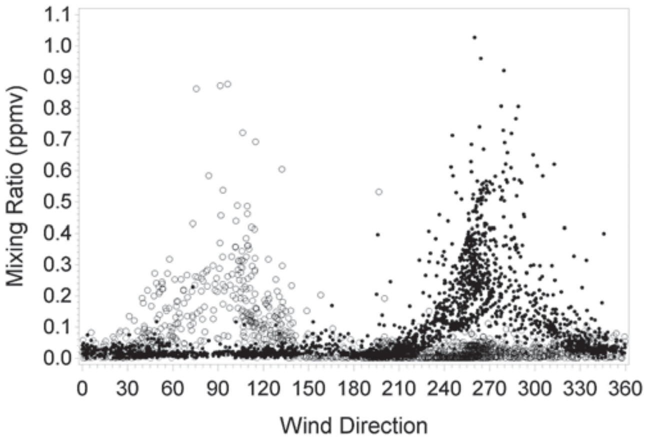

Fig. 2. Ammonia mixing ratio (ppm_v) for the eastern open-path laser (dots) and the western open-path laser (circles) adjacent the cattle feedlot.

temperature using the new manufacturer's calibration curves. On 20 June 2014 (DOY 171), the two NH_3 OPLs and retroreflectors where deployed at the commercial feedlot on the east and west side. The east and west measurement paths (395 and 305 m, respectively; both distances resulted in appropriate OPL return light levels) were oriented parallel to the feedlot's north–south perimeters (Fig. 1). The OPL west path was 1.5 m above the pasture. The OPL east path was adjusted as the canola crop grew and was maintained at approximately 1 m above the crop height. The average NH_3 mixing ratio was recorded along each path every minute. The OPL NH_3 mixing ratio was corrected for air pressure and temperature using the manufacturer's calibration curves. A correction of 0.93 was applied to account for between-OPL differences as derived from the side-by-side OPL comparison ($R^2 = 0.96$).

During the campaign, a three-dimensional sonic anemometer (CSAT3, Campbell Scientific) was mounted 3 m above the soil surface on a tower located 165 m to the east of the feedlot (Fig. 1). The three components of the wind speed were sampled at 10 Hz. A wind vane was used to control sampling of NH_3 concentration at the tower that was used to determine NH_3 flux.

The emission of NH_3 from the feedlot was determined using a dispersion model (WindTrax version 2.0.8.8, Thunder Beach Scientific) that is based on a Lagrangian stochastic procedure (Flesch et al., 2004). The model simulated the release of 10,000 trajectories to characterize the growth of the NH_3 plume from the feedlot. The resulting emission-to-concentration relationship $(Q/C)_{sim}$ was used to infer the NH_3 emission using the OPL data. The background concentration was derived by WindTrax based on the upwind OPL data. Inaccuracy of this method has been evaluated as 10% (Harper et al., 2010) using known tracer release rates.

Land-based Ammonia Deposition and Emission

Ammonia deposition was determined using three approaches: flux gradient (FG), deposition velocity (DV), and soil-capture (SC). Ammonia emission using FG was found only when winds were from the east.

Flux Gradient

At the tower located to the east of the feedlot (Fig. 1), air was sampled at four heights (0.25, 0.5, 1.0, and 2.0 m) above the crop. Air was drawn through a miniature check valve that ensured in-flow only and then through a sorbent tube containing sulfuric acid–coated silica gel beads (226–10–06, SKC) that trapped NH_3. The 0.15 L min^{-1} air flow through each sorbent tube was controlled by a mass flow controller (Alicat Scientific Inc.). The outlets of the controllers were connected to a common inlet of a pump (TD-3L57, Brailsford and Co.).

At each height, solenoids directed the air flow through one of two sorbent tubes or through a bypass inlet. The solenoids were controlled by a datalogger (CR1000, Campbell Scientific) to allow the simultaneous air flow through four

Table 1. Cattle diets over the measurement campaign.

Ingredients	Diet 1	Diet 2	Diet 3	Diet 4	Diet 5	Diet 6	Diet 7
				17 June–9 Oct.			
Corn silage	54†	43	32	21	10	10	21
Barley	41	52	63	74	85	83	60
Starch product	3	3	3	3	3	3	0
Finish supplement	2	2	2	2	2	2	4
Dry distillers grain	0	0	0	0	0	0	15
Ractopamine	0	0	0	0	0	2	0
CP‡ % of DM§	9.96	10.31	10.66	11.02	11.37	11.09	13.45
Ingredients	**Diet 1**	**Diet 2**	**Diet 3**	**Diet 4**	**Diet 5, 7, 8**	**Diet 9**	
				10 Oct–2 Nov.			
Corn silage	54	43	32	21	10	10	
Barley¶	24	35	46	57	68	63	
Starch product¶	5	5	5	5	5	5	
Finish supplement	2	2	2	2	2	2	
Dry distillers grain	15	15	15	15	15	15	
CP % of DM	12.68	13.03	13.4	13.73	14.09	12.63	

† Values are percentage dry matter.

‡ Crude protein.

§ Dry matter.

¶ Between 10 and 15 Oct., starch source was 0%, and barley was 5% greater.

Table 2. Average cattle number, dry matter intake, and nitrogen intake by the cattle.

Interval	Average cattle number	Dry matter intake	Nitrogen intake
		—— kg animal^{-1} d^{-1} ——	
17–23 June	8179	9.9	0.173
24–30 June	8209	9.8	0.177
1–7 July	8031	10.8	0.182
8–13 July	7903	7.9	0.172
14–20 July	8465	8.8	0.172
21–27 July	8421	8.9	0.180
28 July–3 Aug.	8247	9.3	0.183
4–10 Aug.	8012	8.5	0.167
11–17 Aug.	7196	8.5	0.159
1–5 Oct.	8080	9.4	0.159
6–9 Oct.	8103	10.3	0.180
10–15 Oct.†	8422	10.6	0.159
16–20 Oct.	8408	9.5	0.242
21–26 Oct.	8608	8.1	0.187
27 Oct–2 Nov.	8514	7.5	0.164
3–10 Nov.	8919	10.1	0.223
Average ± SD	8244 ± 371	9.2 ± 0.9	0.179 ± 0.022

† Change in diet occurred on 10 Oct.

sorbent tubes, one at each height. The datalogger controlled the start and end time of the air flow through the sorbent tubes and could also bypass the sorbent tube when the wind direction was outside the programmed range. One of two modes was used. In Mode 1, the profile was sampled at two different time intervals where the wind direction was within a single range (e.g., westerly winds). This configuration was useful when the wind direction was steady. In Mode 2, the profile was sampled when the wind direction matched two pre-set wind direction ranges (e.g., easterly or westerly winds) over a single pre-set time interval.

The sorbent tubes were changed each morning, and the operating mode was selected. An extra sorbent tube designated as a "blank" accompanied the exposed sorbent tubes between the laboratory and field. Each tube was flushed with water, and the amount of NH_3 was determined colorimetrically using a manual salicylate method (Kempers and Zweers, 1986).

The NH_3 concentration (C; μg m^{-3}) at each height was derived by multiplying the amount of NH_3 captured in each sorbent tube (minus "blank") by the flow of air through each sorbent tube for the sampling interval. The flux of NH_3 to or away from the crop surface was calculated using the FG technique (Todd et al., 2005). In this technique, the measured vertical NH_3 concentration gradient ($\Delta C/\Delta z$; kg m^{-4}) was multiplied by a transfer coefficient for NH_3 (K_a; m^2 s^{-1}), where K_a was set to equal the transfer coefficient for momentum (K_m) divided by the Schmidt number (0.6) (Flesch et al., 2002):

$$K_m = \frac{k u_* z}{\varphi_m} \quad [1]$$

where k is the von Karman constant (0.4), u_* is the friction velocity (m s^{-1}), z is the measurement height (m), and φ_m is the stability correction. This dimensionless stability correction is defined based on boundary-layer stability (Monteith, 1973):

$$\varphi_m = \left(1 - 16\frac{z}{L}\right)^{0.25} \quad \text{when } \frac{z}{L} < 0 \quad [2]$$

$$\varphi_m = 1 + 4.7\frac{z}{L} \quad \text{when } \frac{z}{L} > 0 \quad [3]$$

where L is the Monin Obukhov length (m).

In our study, $\Delta C/\Delta z$ was derived as the difference in the NH_3 concentration between the highest (2 m) and lowest (0.25 m) sampling height. An exception to this was for deposition, when the 1-m C was the maximum value in the profile. This is speculated to coincide with the 2-m sampler measuring background concentration when the internal boundary layer was below 2 m. The uncertainty in the FG results includes the assumption that the vertical gradient, within an otherwise well-mixed plume, reflects a sink or source at the surface. There is also analytical error that we found to be 3% derived by documenting the recovery of a known release of NH_3 through the sorbent tubes. The FG results also reflect a daytime bias.

Deposition Velocity

The 10-min averaged deposition velocity (v_d; m s^{-1}) was calculated as u_*^2/u (0.04 ± 0.02), which is a maximum that assumes only an aerodynamic limitation exists to the flux of NH_3 (Phillips et al., 2004). Ammonia deposition was derived as the product of the v_d and the C for each 10-min interval, where C (at 1.5 m height) was predicted along an east-west transect (east of the feedlot) using the WindTrax dispersion model. This method is a crude approximation of deposition because deposition affects C differently at different heights. Other uncertainties are related to inaccuracies in the estimates of concentration and feedlot-based emission using WindTrax model (10%) (Harper et al., 2010).

Soil Capture

The net uptake of NH_3 by soil was measured at 40-m intervals on the east side of the feedlot, out to 200 m from the perimeter of the feedlot, over an accumulative interval of typically 7 d. One soil trap was also located on the west side of the feedlot close to the OPL. This was repeated eight times during the study. The SC NH_3 device consisted of a Petri dish holding 20 g of oven-dried soil (Typic Haplustolls) that was positioned close to the ground under a rain shield that was open on all sides. After exposure to air, the soil sample was analyzed. Unexposed soil was the control. Soils were extracted with 2 mol L^{-1} KCl (5 g soil in 25 mL solution) by shaking for 1 h. The NH_3 concentration was determined using an automated colorimetric technique (Bran+Luebbe AutoAnalyzer AA3 Continuous Flow analyzer; method no. G-102–93D). Uncertainties in this approach exist due to inaccuracies in determining the N content of the soil and any N deposited by settling dust that was not measured.

Data Filtering

The feedlot NH_3 emission data (using the dispersion model) were deleted when the OPL return light level was outside the manufacturer's recommended operational range of 2500 to 10,000 (no units) and when the correlation value generated by the OPL was less than 0.96 as used by Flesch et al. (2014).

A second filter was applied that corresponded to atmospheric boundary conditions that are known to violate the underlying assumptions in the dispersion model (Flesch et al., 2014). This second filter eliminated data based on $u_* < 0.05$ m s^{-1} and the difference (>1°) between the measured air temperature gradient (ΔT) and the Monin–Obukhov gradient (ΔT_{MO}):

$$\Delta T_{MO} = \frac{u_*^2\, T_{avg}}{k^2\, g\, L}\left[ln\left(\frac{z_2}{z_1}\right) - \varphi_H\left(z_2\right) + \varphi_H\left(z_1\right)\right] \qquad [4]$$

where T_{avg} is the average air temperature (K, at z = 2 m), g is the gravitational acceleration (9.81 m s^{-2}), and φ_H is the stability correction function given as:

$$\varphi_H = 2\,ln\left[\left(1+\sqrt{1-16z/L}\right)/2\right] \quad \left(\text{for } L<0\right) \qquad [5]$$

$$= -5\frac{z}{L} \qquad \left(\text{for } L>0\right) \qquad [6]$$

Results

Quality Control of Measurements

In applying quality control to the data used to determine the feedlot-based NH_3 emission, the first filter criteria (related to OPL performance) generated 12,374 useable 10-min interval data. Further filtering for invalid meteorological conditions reduced the available 10-min averaged interval data for use in dispersion modeling to 8968 10-min intervals. A final filter applied was the fraction of feedlot area involved in the dispersion model (>0.10 as used by Flesch et al. [2007]), which reduced the number of 10-min intervals used in our study to 1141. Although this final filtering reduced our useable data to 9% of the original data, it represents all downwind measurements (i.e., the larger dataset [8968 data points]).

The sorbent tube NH_3 concentration profile data were examined and quality checked. Of the 177 intervals sampled (177 × 4 tubes = 708 total), 57 were intervals where sampling occurred for the correct wind direction over a minimum of at least 60 min. Of these, 28 intervals showed clear indications of either a deposition or emission concentration profile and were used to characterize NH_3 deposition/emission from the feedlot-adjacent field. The remaining data had an unresolved NH_3 concentration profile due to poor detection at low concentrations.

Ammonia Concentration from Open-Path Lasers and Sorbent Tubes

The NH_3 mixing ratios measured using the two OPL (east and west of the feedlot) varied with the wind direction (Fig. 2). During the entire study, the mixing ratio peaked at about 1.5 ppm$_v$ at the eastern OPL, due to three larger values, but was more typical at about 0.9 to 1.0 ppm$_v$ when the OPLs were directly downwind of the cattle feedlot. When the wind direction was not from the feedlot, the mixing ratio was generally less than 0.05 ppm$_v$, reflecting background concentration.

The sorbent tube NH_3 concentration profile indicated the occurrence of NH_3 deposition (Fig. 3) where the concentration increased with height. The concentrations were much greater for the westerly then easterly winds (e.g., 220–270 μg m^{-3} for westerly and 4–15 μg m^{-3} for easterly winds [Fig. 3]). The greater NH_3 concentration coincides to airflow from the feedlot. There were occurrences when the concentration declined with height above the surface (Fig. 4), implying the emission of NH_3 at the surface. For the case in Fig. 4, the wind direction was easterly; no emissions were recorded for westerly winds. The maximum concentration of 129 μg m^{-3} was found at 0.25 m height, but profile concentration was typically less than 20 μg m^{-3}.

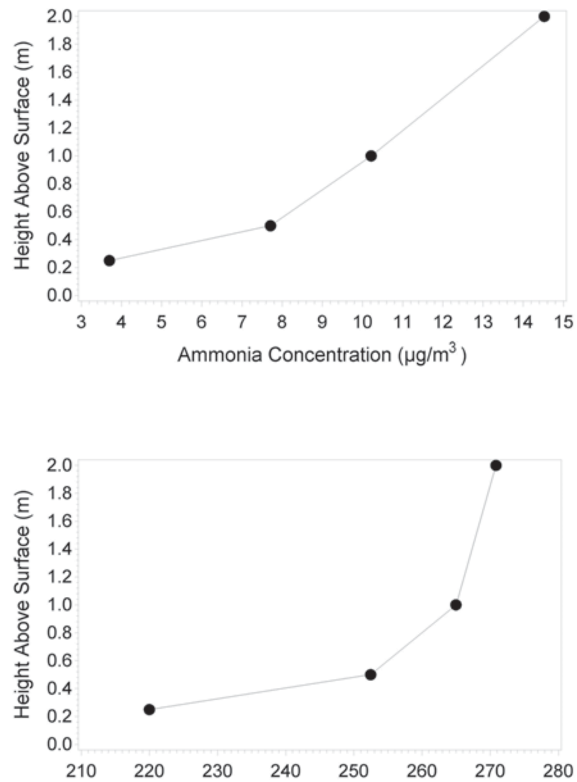

Fig. 3. Concentration profile for two afternoon intervals (1300–1600 h) showing ammonia deposition at the tower location. The wind speed was 4 m s^{-1} for both periods, and the wind direction was 134° (upwind of feedlot; upper curve) and 244° (downwind of feedlot; lower curve).

Soil-Captured Ammonia Deposition

Analysis of soil that was exposed to open air indicated a linear decline in the NH_3 with distance from the feedlot. This is

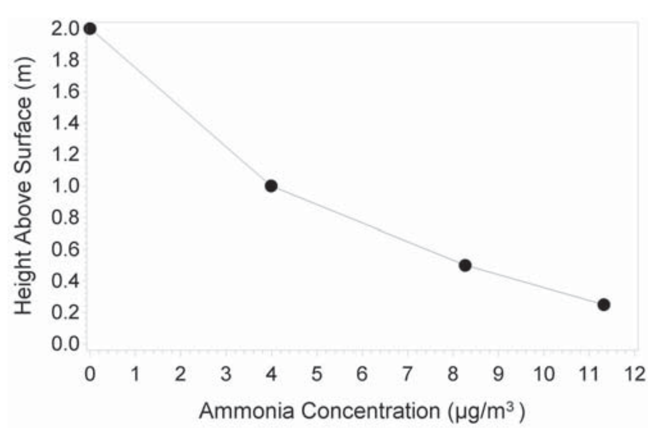

Fig. 4. Ammonia concentration profile at the tower for one afternoon interval (1200 and 1500 h) corresponding to wind speed of 3 m s^{-1} and wind direction of 114° indicating air flow over the crop.

evidence of the horizontal NH_3 gradient on the predominantly downwind side of the feedlot source. On average ($n = 8$), the SC NH_3 rate along the eastern transect varied from 180 mg m^{-2} d^{-1} at a distance of 40 m from the feedlot to 90 mg m^{-2} d^{-1} at a distance of 200 m. On the west side of the feedlot (predominantly upwind), the deposition was 56 mg m^{-2} d^{-1} at the single location. The slope of the deposition with distance from the feedlot was 0.48 mg m^{-2} d^{-1} m^{-1} distance from the feedlot ($R^2 = 0.97$). Linear extrapolation of the relationship indicates zero deposition at 380 m distance from the feedlot.

Feedlot-based NH_3 Emissions

For the feedlot source, the dispersion model calculated an average NH_3 emission of 43 ± 32 μg m^{-2} s^{-1} (73 g animal^{-1} d^{-1}) over the entire campaign. The diel pattern in NH_3 emission from the feedlot (Fig. 5) showed a minimum emission overnight of approximately 25 μg m^{-2} s^{-1} (42 g animal^{-1} d^{-1}) and a peak of 79 μg m^{-2} s^{-1} (135 g animal^{-1} d^{-1}) at mid-day, coinciding with an expected air temperature pattern.

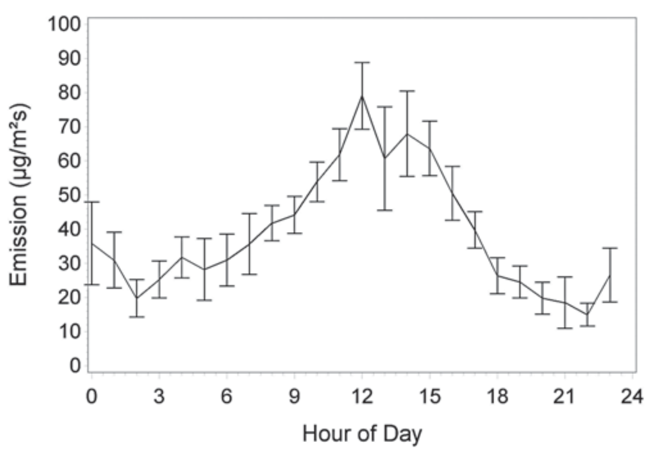

Fig. 5. Hourly averaged ammonia emission at the feedlot using all data. Error bars are 1 SD.

The average feedlot emission of 43 μg m^{-2} s^{-1} was modified because WindTrax does not include deposition. Because deposition acts to reduce the near-surface C (compared with the no-deposition case), using a "reduced" C measurement in WindTrax will lead to an underestimate of the emission rate. Previous deposition modeling work leads us to expect an underestimation of concentration from 10 to 25% (T.K. Flesch, personal communication, October 2015). Despite this limitation, a stepwise integration of the deposition, based on an underestimated emission, indicates a total deposition of 98 kg d^{-1} (7 μg m^{-2} s^{-1}) out to 500 m from the feedlot (Fig. 6). This value was added to the feedlot emission, resulting in a minimum total feedlot loss of 50 μg m^{-2} s^{-1} (85 g animal^{-1} d^{-1}). The feedlot emission when expressed as NH_3–N was 39% of the ingested N by the animal (0.179 kg animal^{-1} d^{-1}) (Table 2).

Land-based Ammonia Deposition and Emission

Using the FG technique in combination with the sorbent tube data, the average deposition of NH_3 was 12.0 μg m^{-2} s^{-1} for westerly winds (air passing over the feedlot) (Table 3). A smaller deposition rate of 5.4 μg m^{-2} s^{-1} was also found for easterly winds (no feedlot source). The magnitude of the concentrations (profile) was much smaller for the easterly deposition events, typically less than 50 ppb (Fig. 3). It follows that deposition to the field adjacent the feedlot occurred from all directions but was greater with air flow from the feedlot.

In addition to the deposition rate varying with the wind direction, it is also expected to differ spatially, coinciding with the near-surface NH_3 concentration. The calculated NH_3 deposition at the tower is unique to the footprint area of the measurements and therefore is not representative of the entire field. The SC data indicate that deposition will be greatest close to the feedlot. This spatial pattern was also found in the DV data, where a maximum deposition of 17 μg m^{-2} s^{-1} at the feedlot perimeter declined to <1 μg m^{-2} s^{-1} at 500 m distance from the feedlot (Fig. 6).

The average field emission was 0.9 μg m^{-2} s^{-1} and was associated with easterly winds at the study site (Table 3). The emissions

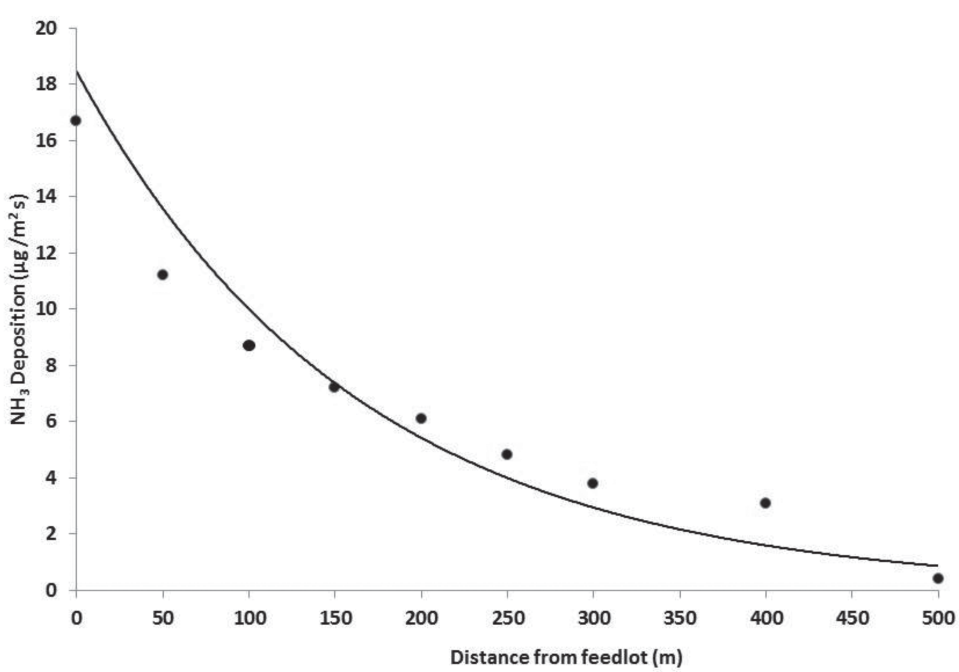

Fig. 6. Maximum deposition for west winds determined from the decline in ammonia concentration at increasing distance from the feedlot.

were associated with very low NH_3 concentrations (<0.01 ppm_v) at the 2-m height (Fig. 4). The same situation is expected for the NH_3 emission from the field (i.e., the emissions are spatially different across the field). However, in this case, the variability is driven by the N content of the surface across the field, which is in part influenced by the NH_3 deposition distribution.

Discussion

The average (minimum) feedlot NH_3 emission of 50 μg m^{-2} s^{-1} (85 g $animal^{-1}$ d^{-1}) is less than the 133 μg m^{-2} s^{-1} reported for typical Australian feedlots (Denmead et al., 2014). Baum and Ham (2009) reported feedlot emissions in Kansas between 68 and 127 μg m^{-2} s^{-1}. Staebler et al. (2009) reported an average emission of 76 μg m^{-2} s^{-1} for a feedlot in the same vicinity as our study. In Texas, cattle feedlot emissions in spring and summer were reported as 90 and 108 μg m^{-2} s^{-1}, respectively (Flesch et al., 2007). In a review of NH_3 emissions from commercial beef cattle feedlots, Hristov et al. (2011) report an annualized emission of 48 μg m^{-2} s^{-1}.

The 39% of the fed N that was lost as NH_3 is similar to the 38% reported by Baum and Ham (2009), less than the 48 ± 15% given by Hristov et al. (2011), and much less than the 60 to 65% reported by Flesch et al. (2007), Todd et al. (2005), and McGinn et al. (2007). Flesch et al. (2007) reported a range for Texas feedlots of 29 to 53%.

Our reported NH_3 emission and percent fed N lost as NH_3 coincides to a lower dietary CP content of 10%, whereas McGinn et al. (2007) reported a CP content of 12.8% and Flesch et al. (2007) reported CP content of about 15%. For an artificial beef feedlot surface in Texas, Todd et al. (2006) reported a 44% decrease in NH_3 emissions associated with CP decrease from 13 to 11.5%. As CP declines in the diet, there is less excess N excreted in manure, and therefore NH_3 emission declines.

There was a strong correlation between peak daily NH_3 emission from the feedlot and peak air temperature. The correlation of NH_3 emission and air temperature coincides with the recognition that higher surface temperature increases the dissociation of NH_4^+ to NH_3–N in manure, which includes urine and produces a greater NH_3 emission.

The decline in NH_3 deposition with distance from the feedlot is expressed in the SC data. The decrease was about 50% (from 2 to 1 μg m^{-2} s^{-1}) at 200 m from the feedlot. The average deposition rate of 1.2 μg m^{-2} s^{-1} at a distance of 165 m (the tower location) was much less than the deposition rate derived using the FG method (12 and 5.4 μg m^{-2} s^{-1} for westerly and easterly winds, respectively) or the DV technique (7 μg m^{-2} s^{-1}). This is expected because the SC data are the accumulated multi-day depositions (from all wind directions), whereas the FG and DV approach results are aligned with short sampling times for specific wind directions. It is speculated that dust deposited to the SC soil sample was not a significant factor. Using the DV results, the deposition of NH_3 out to 500 m distance totaled 97 kg d^{-1}. This deposition was 14% of the total (702 kg d^{-1}) from the feedlot and is similar to the 10% determined by Staebler et al. (2009). The major advantage of the DV approach is that it is easily derived using the WindTrax once the deposition velocity is known.

Table 3. Field ammonia flux (deposition/emission) calculated using a flux gradient technique and sorbent tube accumulation sampling at the tower location (westerly wind places the feedlot upwind).

Wind direction	Flux	Average	SD
		μg m^{-2} s^{-1}	
Easterly	deposition	5.4	3.9
	emission	0.96	0.8
Westerly	deposition	12.0	6.9
	emission	–	–

Conclusions

Our results illustrate the dynamics of reactive NH_3 in the vicinity of a beef cattle feedlot and confirm that a large portion of the fed N as CP is volatized from the feedlot's cattle manure. In the local vicinity of a feedlot, both NH_3 deposition (14% of the emitted NH_3) and emission occurred. The NH_3 deposition was 10-fold that of emission from the field-based exchange. It follows that quantifying the local dry deposition to surrounding fields is required when applying feedlot-based emissions to a large-scale emissions inventory. Future research should focus on using detectors that provide more continuous and short-averaging periods of emissions and deposition.

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References

Arogo, J., P.W. Westerman, A.J. Heber, W.P. Robatge, and J.J. Classen. 2006. Ammonia emissions from animal feeding operations. In: J.M. Rice, D.F. Caldwell, F.J. Humenik, editors, Animal agriculture and the environment. National Center for Manure and Animal Waste Management white papers. ASABE Publ. 913C0306. ASABE, St. Joseph, MI.

Asman, W.A.H., and H.A. van Jaarsveld. 1992. A variable-resolution transport model applied for NH_χ in Europe. Atmos. Environ. 26:445–464. doi:10.1016/0960-1686(92)90329-J

Barthelmie, R.J., and S.C. Pryor. 1998. Implications of ammonia emissions for fine aerosol formation and visibility impairment: A case study from the Lower Fraser Valley, British Columbia. Atmos. Environ. 32:345–352. doi:10.1016/S1352-2310(97)83466-8

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

Bouwman, A.F., D.S. Lee, W.A.H. Asman, F.J. Dentener, K.W. Van Der Hoek, and J.G.J. Olivier. 1997. A global high-resolution emission inventory for ammonia. Global Biogeochem. Cycles 11:561–587. doi:10.1029/97GB02266

Cape, J.N. 2014. What happens to ammonia on leaf surfaces? In: M.A. Sutton, K.E. Mason, L.J. Sheppard, H. Sverdrup, R. Haeuber, and W.K. Hicks, editors, Nitrogen deposition, critical loads and biodiversity. Springer, New York. p. 139–146.

Denmead, O.T., D. Chen, D. Rowell, Z. Loe, J. Hill, S. Muir, D.W.T. Griffith, T. Naylor, M. Bai, F. Phillips, and S. McGinn. 2014. Gaseous nitrogen emissions from Australian cattle feedlots In: M.A. Sutton, K.E. Mason, L.J. Sheppard, H. Sverdrup, R. Haeuber, W.K. Hicks, editors, Nitrogen deposition, criteria loads and biodiversity. Springer, New York. p. 23–29.

Environment Canada. 2014. 2012 air pollutant emission summaries and historical emission trends. www.ec.gc.ca/inrp-npri/default.asp?lang=En&xml=F98AFAE7-9953-410B-9F1A-4ACC980A4882 (accessed 29 Mar. 2016).

Erisman, J.W., and M. Schaap. 2004. The need for ammonia abatement with respect to secondary PM reductions in Europe. Environ. Pollut. 129:159–163. doi:10.1016/j.envpol.2003.08.042

Ferm, M. 1998. Atmospheric ammonia and ammonium transport in Europe and critical loads: A review. Nutr. Cycl. Agroecosyst. 51:5–17. doi:10.1023/A:1009780030477

Flesch, T.K., J.H. Prueger, and J.L. Hatfield. 2002. Turbulent Schmidt number from a tracer experiment. Agric. For. Meteorol. 111:299–307. doi:10.1016/S0168-1923(02)00025-4

Flesch, T.K., J.D. Wilson, L.A. Harper, B.P. Crenna, and R.R. Sharpe. 2004. Deducing ground-air emissions from observed trace gas concentrations: A field trial. J. Appl. Meteorol. 43:487–502. doi:10.1175/1520-0450(2004)043<0487:DGEFOT>2.0.CO;2

Flesch, T., J. Wilson, L. Harper, R. Todd, and A. Cole. 2007. Determining ammonia emissions from a cattle feedlot with an inverse dispersion technique. Agric. For. Meteorol. 144:139–155. doi:10.1016/j.agrformet.2007.02.006

Flesch, T.K., S.M. McGinn, D. Chen, J.D. Wilson, and R.L. Desjardins. 2014. Data filtering for inverse dispersion emission calculations. Agric. Meteorol. 198-199:1–6. doi:10.1016/j.agrformet.2014.07.010

Harper, L.A., T.K. Flesch, K.M. Weaver, and J.D. Wilson. 2010. The effect of biofuel production on swine farm methane and ammonia emissions. J. Environ. Qual. 39:1984–1992. doi:10.2134/jeq2010.0172

Hristov, A.N., M. Hanigan, A. Coe, R. Todd, T.A. McAllister, P.M. Ndegwa, and A. Rotz. 2011. Review: Ammonia emissions from dairy farms and beef feedlots. Can. J. Anim. Sci. 91:1–35. doi:10.4141/CJAS10034

Kempers, A.J., and A. Zweers. 1986. Ammonium determination in soil extracts by the salicylate method. Commun. Soil Sci. Plant Anal. 17:715–723. doi:10.1080/00103628609367745

Loubet, B., W.A.H. Asman, M. Theobald, O. Hertel, S.Y. Tang, P. Robin, M. Hassouna, U. Dämmgen, S. Genermont, P. Cellier, and M.A. Sutton. 2006. Ammonia deposition near hot spots: Processes, models and monitoring methods. In: A. Sutton, S. Reis, and S.M.H. Baker, editors, Atmospheric ammonia. Springer, Dordrecht, The Netherlands. p. 205–267.

MacVean, D.W., D.K. Franzen, T.J. Keefe, and B.W. Bennett. 1986. Airborne particle concentration and meteorologic conditions associated with pneumonia incidence in feedlot cattle. Am. J. Vet. Res. 47:2676–2682.

McCubbin, D.R., B.J. Apelberg, S. Roe, and F. Divita, Jr. 2002. Livestock ammonia management and particulate-related health benefits. Environ. Sci. Technol. 36:1141. doi:10.1021/es010705g

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. 36:1585–1590. doi:10.2134/jeq2007.0167

Monteith, J.L. 1973. Principles of environmental physics. Edward Arnold Ltd, London.

National Research Council. 2000. Nutrient requirements of beef cattle. 7th revised ed. National Academy Press, Washington, DC.

Park, R.J., D.J. Jacob, B.D. Field, R.M. Yantosca, and M. Chin. 2004. Natural and transboundary pollution influences on sulfate-nitrate-ammonium aerosols in the United States: Implications for policy. J. Geophys. Res. Atmos. 109:1–17. doi:10.1029/2003JD004110

Phillips, S.B., S.P. Arya, and V.P. Aneja. 2004. Ammonia flux and dry deposition velocity from near-surface concentration gradient measurements over a grass surface in North Carolina. Atmos. Environ. 38:3469–3480. doi:10.1016/j.atmosenv.2004.02.054

Popendorf, W., K.J. Donham, D.N. Easton, and J. Silk. 1985. A synopsis of agricultural respiratory hazards. Am. Ind. Hyg. Assoc. J. 46:154–161. doi:10.1080/15298668591394572

Sanderson, M.G., W.J. Collins, C.E. Johnson, and R.G. Derwent. 2006. Present and future acid deposition to ecosystems: The effect of climate change. Atmos. Environ. 40:1275–1283. doi:10.1016/j.atmosenv.2005.10.031

Staebler, R.M., S.M. McGinn, B.P. Crenna, T.K. Flesch, K.L. Hayden, and S.-M. Li. 2009. Three-dimensional characterization of the ammonia plume from a beef cattle feedlot. Atmos. Environ. 43:6091–6099. doi:10.1016/j.atmosenv.2009.08.045

Sutton, M., C.E. Pitcairn, and D. Fowler. 1993. The exchange of ammonia between the atmosphere and plant communities. Academic Press, Cambridge, MA.

Todd, R.W., N.A. Cole, L.A. Harper, T.K. Flesch, and B.H. Baek. 2005. Ammonia and gaseous nitrogen emissions from a commercial beef cattle feedyard estimated using the flux-gradient method and N:P ratio analysis. In: O.J. Nowak, editor, Proceedings of State of the Science: Animal manure and waste management. National Center for Manure and Animal Waste Management, San Antonio, TX.

Todd, R.W., A. Cole, and R.N. Clark. 2006. Reducing crude protein in beef cattle diet reduces ammonia emissions from artificial feedyard surfaces. J. Environ. Qual. 35:404–411. doi:10.2134/jeq2005.0045

Voorburg, J., and W. Kroodsma. 1992. Volatile emissions of housing systems for cattle. Livest. Prod. Sci. 31:57–70. doi:10.1016/0301-6226(92)90052-6