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Ammonia emissions from the field application of liquid dairy manure after anaerobic digestion or mechanical separation in Ontario, Canada

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ABSTRACT

Anaerobic digestion (AD) and solid-liquid separation (SLS) are manure treatment strategies used on dairy farms. These treatments change manure characteristics, such as pH, dry matter, and total ammoniacal nitrogen (TAN) which affect ammonia (NH_3) emissions after field application. This study used eight wind tunnels (1 m^2 each) to compare the effects of application season and dairy manure treatment (AD vs SLS) on NH_3 emissions. The highest cumulative NH_3 emissions were seen in the spring from the AD manure treatment, which had higher TAN and pH compared to SLS. On the other hand, when normalized by TAN application rate there was no significant difference between AD and SLS manure in either spring (52 and 54% of TAN after 22 d, respectively) or fall (26% and 27% of TAN after 15 d, respectively). This suggests that if farmers adjusted application rates based on TAN content, NH_3 emissions from applied AD manure could be minimized. Results were compared to the Ammonia Loss from Field Applied Manure (ALFAM) model. The model characterized the temporal pattern of emissions reasonably well in spring, but not fall. The model generally overestimated NH_3 emissions during the first 24 h after application, and underestimated emissions after incorporation from day 2 to 22.

1. Introduction

Ammonia (NH_3) is a toxic gas with serious health and environmental impacts. It contributes to eutrophication, forest dieback and soil and water acidification. In Canada, the agricultural sector is estimated to be responsible for over 90% of NH_3 emissions (Environment and Climate Change Canada, 2016), and 30–50% are directly related to the field application of livestock manure (Bittman and Mikkelsen, 2009; Sintermann et al., 2012). The NH_3 emissions from field-applied manure begin immediately following application. The magnitude of emission rates are dependent on field conditions and manure characteristics including pH, temperature, dry matter (DM) and total ammoniacal nitrogen (TAN) concentration. These characteristics are altered by manure treatments prior to field application.

Anaerobic digestion (AD) and solid-liquid separation (SLS) are manure treatments increasingly used on dairy farms, due to advancing technologies, growing herd sizes and economic incentives (Husfeldt et al. 2012, Harper et al., 2010). Anaerobic digestion has many benefits, including the production of methane as a fuel source, reduction in pathogen populations (Kearney et al., 1993; Côté et al., 2006), and reduced greenhouse gas emissions and odor (Clemens et al., 2006; Harper et al. 2010; Massé et al., 2011; VanderZaag et al. 2017). Alternatively, AD increases the TAN concentration in the effluent, resulting in the potential for increased NH_3 volatilization following field application of digested manure. Most dairy operations with AD systems also use SLS to extract the solid fraction which is used for bedding material (pathogen free), and apply the separated liquid fraction to fields after storage (Husfeldt et al. 2012).

Potential NH_3 emissions due to use of AD can increase in two ways: (i) AD facilitates the mineralization of organic N (Field et al., 1984); and (ii) AD increases the temperature and pH, forcing the NH_3-NH_4 equilibrium towards the volatile NH_3 (Fillingham et al., 2017). Amon et al. (2006) suggested that this equilibrium change could result in increased NH_3 emissions from digested manure after field application. They found a significant increase of 18% in NH_3 emissions in summer-pasture applied AD dairy manures compared to untreated manure in the

Abbreviations: NH_3, ammonia; GHG, greenhouse gases; AD, anaerobic digestion; SLS, solid-liquid separation; TAN, ammoniacal nitrogen

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48 h immediately following application. However, Clemens et al. (2006) found no difference in cumulative NH_3 emissions from AD and untreated beef manure 8 d after application. In studying swine manure, Chantigny et al. (2004, 2007 and 2009) suggested that within the first 24 h of application, NH_3 emissions from AD manure were higher than from untreated manure. Differences were neutralized or reversed after the first 24 h, resulting in either no difference in cumulative NH_3 emissions (Chantigny et al., 2004), or a decrease from AD manure after 1 day for spring applied manures and higher overall emissions from untreated fall applied manures (Chantigny et al., 2007 and 2009). Both Clemens et al. (2006) and Chantigny et al. (2004) attributed equal or lower AD emissions to more rapid soil infiltration, due to lower total solids content and decreased viscosity in AD manure. Sun et al. (2014) found that AD reduced total solids up to 50%.

Mechanical solid-liquid separation is used independently and in conjunction with AD. Sun et al. (2014) compared NH_3 emissions from both AD and untreated dairy manures with and without solid-liquid separation. Their findings on AD were inconsistent, but they found higher solids content and TAN concentrations led to higher NH_3 emissions. The applicability to other locations is unclear since the research was conducted during the summer in a very dry location (Puyallup, Washington, USA). In Canada, most dairy manure is applied in spring and fall (65%; Beaulieu 2004). To date, no Canadian published study has focused on spring and fall NH_3 emissions from field applied dairy manure treated by AD and SLS.

The objective of this study was to compare the effects of application season and dairy manure treatment (AD vs. SLS manure) on NH_3 emissions from field application. Additionally, our results will be compared to the Ammonia Loss from Field Applied Manure (ALFAM) model, which is an empirical model developed by Søgaard et al. (2002) and has been used in NH_3 assessments in Canada (Sheppard et al., 2011) but requires further testing for simulating emissions from treated manure.

2. Materials and methods

2.1. Experimental site & setup

The experiment was conducted on a Grenville series sandy loam field site at the Central Experimental Farm of Agriculture and Agri-Food Canada in Ottawa, Ontario (45° 38′, 75° 71′, altitude: 79 m). The field was chosen for its proximity to a power supply and meteorological station. The 1981–2010 historical annual mean temperature for the area is 6 °C with an average annual precipitation of 943.5 mm (Environment Canada, 2014). Prior to the study, a composite soil sample (0–15 cm) from the field was analyzed at SGS Agri-Food Laboratories (Guelph, Ontario; Table 1).

Eight 1-m^2 plots were established in the fall of 2013, and another eight in spring 2014. Two of the plots were controls, receiving distilled water. Three replicates of each manure treatment (AD and SLS) were randomly assigned to the remaining six plots. Manures and distilled water were applied evenly at equivalent rates of 7 L m^{-2} (70,000 L ha^{-1}), on their respective plots. Manures were incorporated 24 h after application with a shovel to a depth of 15 cm.

Table 1
Soil properties before the experiment began.

Parameter	Soil Properties
pH	6.9
Total N (%)	0.2
Total Org C (%)	1.8
Phosphorus (ppm)	34.6
Potassium (ppm)	210.3
Magnesium (ppm)	148.3
% Sand	58
% Silt	31
% Clay	11
CEC (cmol(+) kg^{-1})	16.5

2.2. Ammonia volatilization measurements

Immediately after manure treatments were applied, the plots were covered by wind tunnels to measure NH_3 emission rates. In total there were eight wind tunnels, similar to those described by Lockyer (1984), Rochette et al. (2009) and Smith et al. (2007). Tunnels were comprised of a rectangular, steel frame (2m × 0.5 m) covered by a clear plexi-glass dome over each 1-m^2 plot. The tunnels attached to a steel duct containing a fan drawing air over the plot. Tunnel wind speed was measured daily by a hot-wire anemometer (407123-NIST, Extech Instruments, Boston) through a small port on the top of the duct, where the typical wind speed was 12 m s^{-1}. The tunnel airflow rate was typically 10 m^3 s^{-1}, providing an air-exchange rate of at least 22 exchanges per minute. Air was drawn from a sampling port in the duct to a 0.005M H_3PO_4 acid trap and through a gas flow meter that measured the total flow during each sampling interval. The average flow rate was set by a critical orifice at 0.5 L min^{-1} in the fall. Orifices were replaced during the winter to increase sample flow to 3 L min^{-1} in the spring. In the fall, sampling was done over 15 days, and stopped to prevent damage to glassware when air temperatures were forecast to drop well below freezing. In the spring, sampling continued over 22 days.

Acid traps were sampled daily for the first 8 d, and every other day for the remainder of the experimental period. Acid samples were collected into 125 mL tubes and frozen until analysis. Samples were analyzed using the Lachat QuikChem Flow Injection Analysis (FIA) 8000 and QuikChem method 13-107-06-2-D (Lachat Instruments, Ltd, Loveland, CO).

2.3. Manure source and collection

Samples of AD and SLS were obtained from two commercial dairy farms. The AD manure was taken from a 140 head dairy farm in southwestern Ontario where excreted manure was scraped from the barn floor, mixed with off-farm material (e.g., liquid waste from cookie manufacturing and solid fruit and vegetable waste from a food processing facility), and then added to a 2-stage mesophilic anaerobic digester. After digestion, digestate passed through a mechanical screw press (DariTech EYS separator, Lynden, WA). The SLS manure was taken from a 160 head dairy farm in eastern Ontario where manure was scraped from alleys and then separated by a mechanical screw press (DariTech EYS separator, Lynden, WA). Both farms used milking robots and manure solids for bedding.

At both farms, 40 L of manure was collected prior to each application trial as a composite from around the perimeter of manure storages using a column sampler that extended the entire depth of the tank. Buckets were sealed with a lid and refrigerated until application. All manure was stored for fewer than 3 d prior to application. The pH and TAN concentrations were consistently higher in the AD manure than the SLS manure (Table 2).

At the time of application, a sample was taken by thoroughly mixing all stored manure and taking 0.5 L from each bucket for a total of 1 L. Each sample was analyzed for pH, total N, NH_3-N and DM content. The amount of N applied to each tunnel was calculated as the product of the total N concentration and the volume of manure applied.

2.4. Environmental measurement and rainfall simulation

Meteorological data, including ambient temperature, precipitation, solar radiation, and sub-surface soil temperatures were obtained from a nearby weather station (Ottawa CDA, Climate ID 6105976). Soil surface temperature and soil moisture content in each tunnel plot were measured each sampling day. Soil surface temperature was measured at

Table 2
Properties of applied anaerobic digestate (AD) and solid-liquid separated (SLS) manure in each season. Analytes include pH, total ammoniacal nitrogen (TAN), TAN application rate to the plots (kg TAN ha^{-1}), total nitrogen (TN), N application rate (kg TN ha^{-1}), and dry matter content (DM).

	Spring 2014		Fall 2013	
	AD	SLS	AD	SLS
pH	8.1	7.1	7.7	7.6
TAN (ppm)	2,600	730	1,600	904
TAN Application Rate (kg N ha^{-1})	182	51	112	63
Total N (%)	0.36	0.17	0.20	0.13
N application rate (kg N ha^{-1})	252	119	140	91
DM (%)	3.0	3.8	2.7	2.0

three locations inside each tunnel using an infrared thermometer (15-077-967, Traceable, Fisher Scientific, Ottawa, ON). Soil moisture was measured by inserting a probe (FieldScout TDR 100 Soil Moisture Meter, Spectrum Technologies, Aurora, IL) into the soil within each tunnel at three locations and averaged. Since the tunnels were covered, equivalent rainfall was applied manually using a watering can after each rain event.

2.5. Data analysis

Aqueous NH_3-N concentrations were converted to gaseous concentrations using the following:

$$C_{air} = \frac{C_{aq} \times V_{aq}}{V_{air}} \tag{1}$$

where C_{air} (mg NH_3-N m^{-3}) is the NH_3-N concentration in the air sampled using the wind tunnel, C_{aq} (mg NH_3-N L^{-1}) is the aqueous measured NH_3-N concentration, V_{aq} (L) is the volume of acid in the sample and, Vair (m^3) is the volume of air passed through the sample during the sampling time.

The NH_3-N concentration in the sample air (C_{air}) was converted to a flux (F, mg m^{-2} s^{-1}) using the following:

$$F = \frac{Q}{A}(C_{air} - C_b) \tag{2}$$

where Q (m^3 s^{-1}) is the wind tunnel air flow rate, calculated by multiplying the measured wind speed in the duct by the cross-sectional area of the duct, A is the surface area (1 m^2), and C_b is the background NH_3-N concentration (mg m^{-3}).

The average fluxes from control tunnels were subtracted from the manure tunnel fluxes. Control tunnels were run with only water under the same conditions as tunnels where manure/digestate was applied. Fluxes were then converted to mass of N lost over the sampling period by multiplying by the amount of sampling time (s) and the plot area (m^2). The cumulative emissions from each tunnel were taken as the sum of the daily mass emissions by day.

To compare the two treatments, a two-tailed, paired t-test was performed on the cumulative NH_3–N emissions from the three treatment replicates and the normalized emissions from treatment replicates. Results were normalized by the amount of TAN applied and N applied on each treatment by dividing the cumulative emissions of each replicate by the total TAN or N applied. The total TAN or N applied was calculated by multiplying the TAN or N concentration in the composite manure sample by the volume of manure applied.

2.6. Modelling

Using the ALFAM statistical regression model developed by Søgaard et al. (2002), measured NH_3 emissions were compared to model estimates. The model uses a Michaelis-Menten type equation to predict cumulative NH_3 loss over time:

$$N(t) = N_{max}\frac{t}{t + K_m} \tag{3}$$

where the dimensionless *N(t)* is the ratio of cumulative NH_3 loss at time (t) (kg N ha^{-1}) over the N application rate (kg N ha^{-1}). N_{max} is the ratio of cumulative NH_3 loss over time (kg N ha^{-1}) over the N application rate (kg N ha^{-1}) as time approaches infinity. The parameter K_m is the time (h) when *N(t)* is ½N_{max}.

Both N_{max} and K_m are a function of soil water content, air temperature, wind speed, manure type (cattle or swine), manure DM content, TAN content, application method, application rate, incorporation and measurement technique and are fit to the model using a database of almost 6000 records from seven European countries (Søgaard et al., 2002). The model does not include manure pH as an independent variable but rather assumes a value based on slurry type (pH of 7.34 for Cattle slurry).

The ALFAM model was used to estimate NH_3 emissions each season, using environmental conditions (temperature and wind speed) obtained from the nearby meteorological station and the DM and TAN content of both dairy manure types. The model includes a *slurry incorporation* option, which can be either no incorporation or immediate shallow cultivation. In our field trial, incorporation occurred 24 h after application, which is not one of the ALFAM options. Therefore, the model was run twice for these seasons. First, the model simulated 24 h with *no incorporation* and the initial TAN concentration. The modeled N loss over 24 h was then subtracted from the TAN and N application rates. The model was then run a second time with *shallow incorporation* for the remainder of the experiment.

3. Results and discussion

3.1. Environmental conditions

During the spring application trial (22 d starting on May 22, 2014), the average air temperature was 18 °C (Fig. 1), which was similar to the 1981 to 2010 average for the area (13.5 °C for May and 18.1 °C for June; Environment Canada 2014). The daily average precipitation was 2 mm d^{-1} (44.8 mm) slightly lower than the historical average for May of 2.8 mm d^{-1} and June of 3.1 mm d^{-1} (65.2 mm). Sub-surface soil temperature decreased with increasing soil depth during the spring, and soil surface temperature measured with an infrared thermometer reached over 45 °C at times of high solar radiation. While this could suggest the wind tunnels had a warming effect on soil temperature, previous studies with a similar tunnel design having a high air exchange rate suggest temperature differs < 1 °C compared to ambient conditions (Ryden and Lockyer, 1985; Sommer and Olesen, 1991). High air exchange rates are also desirable to avoid inhibiting NH_3 emissions. Rhoades et al., (2005) suggest at least 1.0 air exchange per minute, and results of other studies suggest at least 10–15 exchanges per minute (Cole et al., 2007; Parker et al., 2013). Compared to micrometeorological techniques, wind tunnels may exhibit an oasis effect (observed with pan evaporation by Parker et al., 2013); however, Watt et al. (2016) did not observe a measurable oasis effect for NH_3 flux. A review of NH_3 emissions by Sintermann et al. (2012) indicated that NH_3 fluxes from wind tunnel studies were lower than mid-size micrometeorological methods (usually the Integrated Horizontal Flux method) and higher than field-scale micrometeorological techniques (inverse dispersion or mass-balance methods).

In the fall, application occurred for 15 d starting on October 25, 2013 when average daily air temperatures were below 0 °C on several occasions. There were two markedly warmer days on days 8 and 14 (Fig. 1). The average air temperature for the trial was 3.9 °C and precipitation averaged 2.0 mm d^{-1} (34.2 mm). The daily historic average for the month of November is 2.0 °C and daily precipitation is 2.8 mm (~47.6 mm) (Environment Canada, 2014). Soil temperatures were generally warmer with increasing depth except for days 8 and 14 when

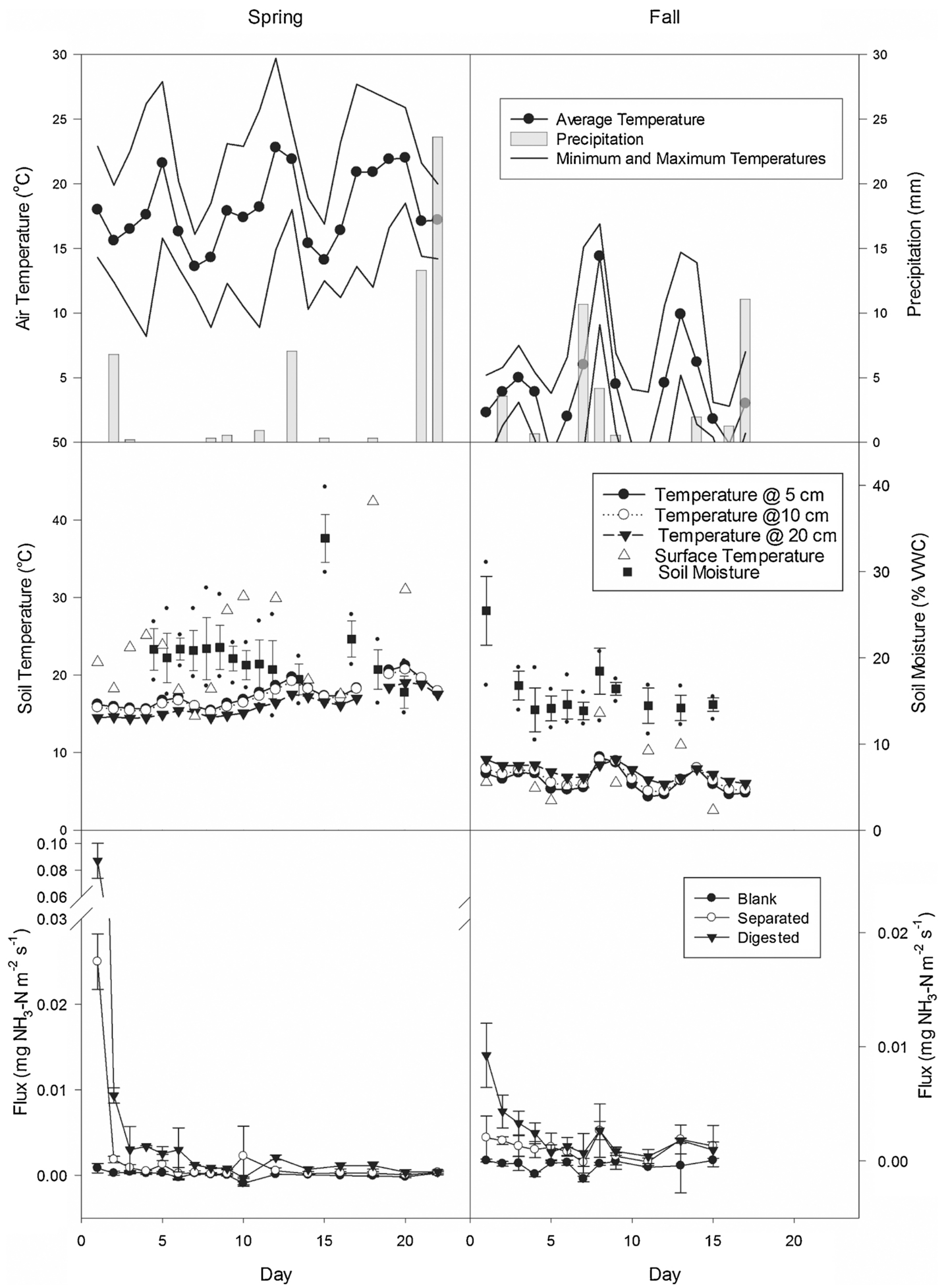

Fig. 1. Top: Daily average, minimum and maximum air temperatures (°C), and precipitation (mm) from Spring, May 22, 2014 to June 12, 2014 (left) and Fall, October 25 2013 to November 8 2013 (right). Middle: Daily sub-surface soil temperatures measured at 5, 10 and 10 cm (°C) and surface temperature measured from Spring (left) and Fall (right) Bottom: Average daily fluxes from each treatment in mg NH_3-N m^{-2} s^{-1} from Spring (left) and Fall (right).

warm ambient temperatures warmed the soil. Fall soil surface temperatures within the tunnels peaked at 16 °C on day 8, corresponding to peak sub-surface temperatures.

Soil moisture had a large increase in the spring trial on day 14, following rainfall on day 13. Rain on days 7 and 8 in the fall corresponded to an increase in the soil moisture on day 8.

3.2. Ammonia fluxes

In the spring, NH_3 fluxes from the control tunnels remained near zero for the duration of the experimental period (Fig. 1). Peak emissions for both treatments occurred in the first day with a rapid decrease after incorporation on the second day. The SLS fluxes were significantly higher ($p = 0.02$) than the control while fluxes from the AD manure were higher than both SLS and the control ($p < 0.05$). In the fall, NH_3 fluxes were significantly lower than spring due to cooler temperatures and fluxes from the control tunnels were near zero (Fig. 1). Fluxes from AD manure were higher than SLS manure and control treatments after the first 24 h. Two small increases in fluxes occurred on day 8 and 13 when the temperatures spiked to over 10 °C.

Fluxes responded to variations in temperature and precipitation. In both seasons and both treatments recorded peak emissions during the

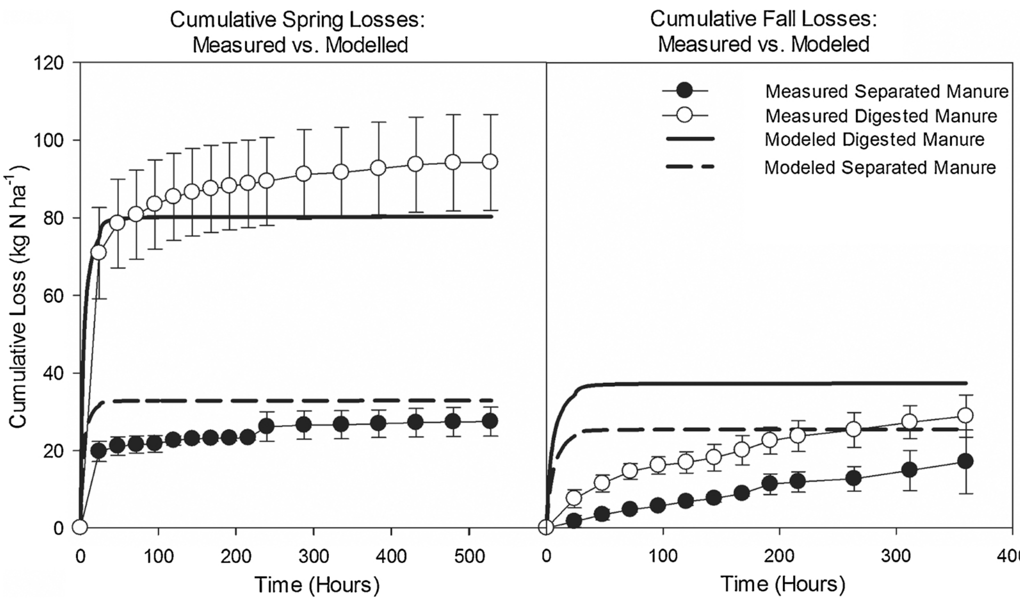

Fig. 2. Measured cumulative NH_3-N emissions (kg N ha^{-1}) from applied anaerobic digestate, and separated dairy manure (liquid fraction) and modeled loss (kg N ha^{-1}) by the ALFAM model for the corresponding manure properties and environmental conditions in the spring (left) or fall (right).

Table 3
Cumulative ammonia emissions normalized by applied N and TAN.

Season	Treatment	Normalized emissions (% of N applied)				Normalized emissions (% of TAN applied)			
		Day 1	Day 2	Day 7	Last Day	Day 1	Day 2	Day 7	Last Day
Spring (22 d)	AD	28.1	31.2	4.7	34.7	38.9	43.1	48.0	51.8
	SLS	16.6	17.8	9.4	23.0	38.7	41.4	45.2	53.8
	(p value)	0.075	0.053	0.026	0.031	0.976	0.771	0.577	0.693
Fall (15 d)	AD	5.4	8.2	14.4	20.7	6.7	10.3	18.0	25.8
	SLS	1.8	3.7	9.8	18.8	2.6	5.4	14.0	27.1
	(p value)	0.119	0.092	0.071	0.685	0.152	0.130	0.132	0.848

Table 4
Comparison of ALFAM model results and cumulative measured NH_3 emissions for the full measurement period.

Season	Treatment	Measured Loss		Modeled Loss		Difference (%)
		kg N ha^{-1}	% of TAN Applied	kg N ha^{-1}	% of TAN Applied	Modeled - Measured
Spring (22 d)	SLS	27.5	53.8	32.9	64.4	16.5
	AD	94.3	51.8	80.3	44.1	−17.5
	% Difference	−242.9	3.7	−144.1	31.5	
Fall (15 d)	SLS	17.2	27.1	22.8	36	24.7
	AD	28.9	25.8	37.4	33.4	22.8
	% Difference	−68.0	4.8	−64.0	7.2	

first 24 h, regardless of incorporation. All applications and treatments continued to have fluxes significantly ($p < 0.1$) above the control plots on more than half of the sampling days until the end of the trials. Spring fluxes in the first 24 h from the AD manure were the highest of all treatments and seasons, which is consistent with the warmer temperatures combined with spring AD manure having the highest TAN concentration, N concentration and pH of all manures used in the experiment.

3.3. Cumulative ammonia emissions

Twenty-two days after application in the spring, the cumulative NH_3 emissions were 9.5 g m^{-2} (95 kg N ha^{-1}) from the AD manure, 2.8 g m^{-2} (28 kg N ha^{-1}) from the SLS manure and 0.9 g m^{-2} (9 kg N ha^{-1}) from the control tunnels. In the fall, after 15 d the cumulative emissions were 2.6 g m^{-2} (26 kg N ha^{-1}) from the AD manure, 1.5 g NH_3-N m^{-2} (15 kg N ha^{-1}) from the SLS manure and 0 g m^{-2} from the control tunnels. In both seasons, it is worth noting that NH_3 emissions slowed after the first 24 h but did not stop completely despite incorporation (Fig. 2). Cumulative NH_3 emissions were highest in the spring from the AD manure, emissions continued to increase after the first 24 h and incorporation. Spring emissions were high from the digested manure due to the high TAN content applied in the spring. The lowest cumulative emissions were observed in the fall when cold temperatures and incorporation limited the potential for loss. The trend at the end of the 15 d in fall is still upwards for both manures (Fig. 2) suggesting that, if above freezing conditions were maintained, loss would continue past the 2 weeks.

3.4. Normalized NH_3 emissions

To compare across seasons, normalized emissions were compared based on the initial 7 d after application. In the spring trial, NH_3 emissions from SLS application resulted in a loss of 23.0% of applied N and of 34.7% from AD. Normalized by TAN, there was little difference between manure types over 7 d. Spring applied SLS lost 45.2% of TAN while AD lost 48.0% of TAN. In the fall, SLS lost 9.8% of applied N and 14.0% of applied TAN. In comparison, fall losses from AD were higher at 14.4% of applied N and 18.0% of TAN.

A significant manure treatment effect was found in the spring when normalized by applied N after 7 d and after the full 22 days of the trial (Table 3). In the fall, much lower loss rates were observed for both manure types due to lower temperatures. When normalized by TAN applied there were no significant treatment effects at any stage in the experiment. This suggests that the greater NH_3 loss from AD compared to SLS manure is simply due to the greater proportion of TAN in the AD manure, because of the digestion process having broken down organic N into TAN.

3.5. Comparison to ALFAM model

The model characterized the temporal pattern of spring NH_3 emissions reasonably well. For spring applied AD, the model predicted total emissions of 80.3 kg NH_3-N ha^{-1} which was lower than measured emissions of 94.3 kg NH_3-N ha^{-1} (Fig. 2). Spring AD manure had the highest pH, which was not considered in the model and likely contributed to the underestimation. The model overestimated emissions from SLS manure, therefore predicting a much larger treatment difference (31.5%) than observed (3.7%) (Table 4). Whereas modeled emissions mostly occurred on the first day, measured emissions continued through day 21 (Fig. 2). At day 2, modeled and measured emissions for AD in spring were similar, but became significantly different later on (Fig. 2). The model also overestimated loss from SLS on the first day but the difference narrowed with time.

Cumulative fall emissions were over-estimated by the model (Table 4), and the temporal pattern was poorly represented (Fig. 2). Emissions of NH_3 were greatly over-estimated in the first 24 h, while emissions later in the experiment were underestimated. Previous field trials have also shown that treatment differences become more apparent after the first 24 h (Gordon and Schuepp, 1994). Cumulative emissions were still increasing after 15 d, and were approaching the total loss indicated by the model. The pH of both separated and digested manure were higher than the model average (7.3) at 7.6 and 7.7, respectively. Although the model overestimated cumulative fall NH_3 emissions, it accurately predicted the difference between the two manures: 64% difference in emissions, and 7.2% difference normalized by TAN applied.

3.6. Comparison to previous studies

Misselbrook et al. (2000) suggested that 37% of applied TAN would be lost after application for cattle slurries applied in the spring and fall. In the fall, loss rates from both the AD and SLS manures were much lower at 25.8% and 27.1% of TAN. More rapid infiltration rates resulting from separation and colder temperatures during the Canadian fall could account for the difference between the results of the present study and those of Misselbrook et al. (2000). Spring emissions were much higher than the suggested 37% at 53.8% and 51.8%, which are closer to the 60% suggested for the summer months by Misselbrook et al. (2000). Warm high temperatures during the spring application were more representative of a summer period.

The lack of significant treatment effect after normalizing in the fall is consistent with Chantigny et al. (2009), where cooler temperatures resulted in lower emissions from digested swine manures. The lack of treatment effect is likely the result of cool and occasionally freezing temperatures of the fall application combined with a smaller difference in pH between the two manures and a smaller differential in NH_3-N concentration between the two manures (compared to the spring). In the spring, the treatment effect was only significant after 24 h, whereas Chantigny et al. (2009) observed continuously high emissions because there was no incorporation. Without incorporation, emissions from untreated manure (with higher DM) continued long after digested manure had infiltrated into the soil.

4. Conclusions

Overall, peak emissions for both treatments occurred in the first day with a rapid decrease after incorporation on the 2nd day. The highest cumulative NH_3 emissions were seen in the spring from the AD manure treatment, which had the highest TAN concentrations and pH. Normalized by TAN application there was no significant difference between AD and SLS manure in either spring or fall. Thus, farmers can avoid high NH_3 emissions from AD manure by adjusting application rates based on TAN content.

Considering its simplicity, the ALFAM model performed quite well. However, it generally overestimated NH_3 emissions, especially in the first 24 h. It also underestimated the influence of NH_3 emissions after incorporation. Including pH in the model would likely improve the ability to model emissions from treated manures.

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