Gaseous Nitrogen Emissions and Forage Nitrogen Uptake on Soils Fertilized with Raw and Treated Swine Manure

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Treatments to reduce solids content in liquid manure have been developed, but little information is available on gaseous N emissions and plant N uptake after application of treated liquid swine manure (LSM). We measured crop yield, N uptake, and NH_3 and N_2O losses after the application of mineral fertilizer (NH_4NO_3), raw LSM, and LSM that was decanted, filtered, anaerobically digested, or chemically flocculated. The experiment was conducted from 2001 to 2003 on a loam and a sandy loam cropped to timothy (*Phleum pratense* L.) with annual applications equivalent to 80 kg N ha^{-1} in spring and 60 kg N ha^{-1} after the first harvest. Raw LSM resulted in NH_3 emissions three to six times larger ($P < 0.05$) than mineral fertilizer. The LSM treatments reduced NH_3 emissions by an average of 25% compared with raw LSM ($P < 0.05$). The N_2O emissions tended to be higher with raw LSM than with mineral fertilizer. The LSM treatments had little effect on N_2O emissions, except for anaerobic digestion, which reduced emissions by >50% compared with raw LSM ($P < 0.05$). Forage yield with raw LSM was >90% of that with mineral fertilizer. The LSM treatments tended to increase forage yield and N uptake relative to raw LSM. We conclude that treated or untreated LSM offers an alternative to mineral fertilizers for forage grass production but care must be taken to minimize NH_3 volatilization. Removing solids from LSM by mechanical, chemical, and biological means reduced NH_3 losses from LSM applied to perennial grass.

SWINE production is concentrating on fewer and larger farms in several countries, resulting in increased manure loading rates and overfertilization of soils. Separation of liquid animal manure into dry matter (DM)-rich and clarified liquid fractions is proposed as an option to improve manure nutrient management and avoid overfertilization (Pain et al., 1978; Møller et al., 2000). The DM-rich fraction contains most of the initial P and can be exported at a reasonable cost to other areas, whereas the liquid fraction contains most of the initial N and is preferably applied on-farm because of its high water content. Mechanical, chemical, and biological treatments for the separation or digestion of solids in liquid animal manure affect the distribution of manure nutrients in the liquid fraction (e.g., Pain et al., 1978; Sommer and Husted, 1995; Møller et al., 2000) and can thus affect the cycling of manure N after land application (Kirchmann and Lundvall, 1993; Sørensen and Thomsen, 2005).

Ammonia volatilization is of particular concern after field application of liquid manure and can result in the loss of a substantial portion of applied N. Mechanical separation of liquid manure was found to decrease NH_3 emissions compared with raw manure (Vandré et al., 1997; Amon et al., 2006). This difference was attributed to lower solid content in the separated than in raw manure, which improved its infiltration below the soil surface. However, some studies reported similar NH_3 losses between separated and raw manure (Thompson et al., 1990; Mattila and Joki-Tokola, 2003). Stevens et al. (1992) found that significant reduction in NH_3 losses was achieved by mechanical separation provided that the removal of solids was extensive enough to increase slurry infiltration in soil.

Anaerobic digestion also decreases the solid content of liquid manure and may thus help to reduce NH_3 emissions after field application. Studies dealing with the effect of anaerobic digestion of manure on NH_3 emissions in the field yielded contrasting results, with some authors reporting lower emissions with digested than with raw manure (Rubaek et al., 1996), whereas others found similar (Pain et al., 1990; Wulf et al., 2002a; Chantigny et al., 2004) or higher emissions (Amon et al., 2006). This discrepancy among

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Abbreviations: LSM, liquid swine manure; DM, dry matter.

studies is likely explained by the increase in pH of liquid manure during anaerobic digestion, which increases the potential for NH_3 loss from digested products and can offset the benefit of improved infiltration (Pain et al., 1990; Wulf et al., 2002a).

Significant N_2O emissions usually follow field application of liquid animal manure. The effects of manure treatments on N_2O emissions can also be variable. Some authors reported similar N_2O emissions between treated and raw liquid manure for mechanical separation (Clemens et al., 1997) and anaerobic digestion (Wulf et al., 2002b), whereas others reported decreased emissions after separation (Amon et al., 2006) and anaerobic digestion (Rubaek et al., 1996; Petersen, 1999; Vallejo et al., 2006). Wulf et al. (2002b) reported similar N_2O emissions between anaerobically digested and raw manure when applied to arable soils but higher emissions with the digested manure when applied to grassland soils.

Liquid swine manure (LSM) is rich in mineral N and was found to be an efficient fertilizer for crop production (Burns et al., 1987, 1990; Morris and Lathwell, 2004; Adeli et al., 2005). By modifying N distribution and by affecting gaseous N losses, treatments to remove solids in LSM could influence N availability to plants after land spreading. Plant N uptake was reported to be similar or slightly increased with mechanically separated (Mattila et al., 2003; Sørensen and Thomsen, 2005) and anaerobically digested (Rubaek et al., 1996) than with raw liquid manure.

Because most studies investigated only one type of manure treatment at a time, it is not possible to state whether the discrepancies among studies on the effects of treatments on gaseous N emissions are due to differences in treatment type, manure source, or environmental conditions. Moreover, these studies were generally conducted on dairy cattle manure, and few studies examined the effects of manure treatments on gaseous N emissions and plant N recovery at the same time. The present field experiment aimed at comparing the effects of various treatments to remove solids in LSM on the short-term NH_3 and N_2O losses and crop yield and N uptake after land application of the liquid fraction to two soils cropped to perennial forage grasses.

Materials and Methods

Manure Collection and Analysis

Raw LSM was obtained during winters of 2001, 2002, and 2003 from a commercial finishing hog operation. The animals were fed a corn-soybean based diet and raised on slatted floor with minimal use of bedding. The LSM was collected by composite subsampling during the emptying of the transfer tank so that the composition of the collected LSM reflected the average composition of LSM in the transfer tank. The LSM had been accumulating for about 48 h in the transfer tank at time of collection. The total volume of collected LSM was thoroughly stirred, and part of it was transferred into a cool-temperature, batch anaerobic digester (Massé et al., 1996). The rest was stored for 6 wk in four 1-m^3 plastic containers. Containers were hermetically closed during storage period. After this period, the upper half of raw LSM was pumped out of two plastic containers and transferred in an empty 1-m^3 container. This manure was labeled "Decanted LSM" and

was assumed to represent the clarified fraction of LSM after natural settling of solids at the bottom of a storage tank. Manure from a third plastic container was allowed to drain through a bed of wood shavings and saw dust (50:50, v/v). The filtrate (about 80% of the initial LSM volume) was collected in an empty plastic container and labeled "Filtered LSM." Manure in the fourth plastic container was used as is and labeled "Raw LSM." After standing for 1 mo in the batch digester, anaerobically digested LSM was transferred into a 1-m^3 plastic container and labeled "Digested LSM." A fifth LSM type was obtained from the transfer tank of a second finishing hog operation equipped to remove solids with a Ca-based coagulant. Animal and manure management were similar to those practiced on the first hog operation. Briefly, LSM was pumped from the transfer tank to a treatment tank where coagulant and pressurized air were injected into the manure so that particulate matter aggregated and floated on the top of the tank. Floating material was removed with scrappers, and 1 m^3 of the treated liquid fraction was collected in a plastic container and labeled "Flocculated LSM." Details about coagulant composition and dosage were not available.

The five plastic containers with the various LSM types were transported to the experimental sites and continuously stirred during field applications. On each application date, a 2-L composite sample was collected from each LSM type for analysis. The LSM samples were first homogenized with a Polytron (Model PT 3100; Kinematica AG, Littau-Lucerne, Switzerland), and pH was measured by direct reading with a glass electrode. Dry matter content was determined as the weight of materials remaining after drying 100 mL of LSM at 55°C for 96 h. Total C concentration was measured by injecting homogenized LSM into an automated combustion C analyzer (Model Formacs; Skalar Analytical, De Breda, The Netherlands).

Total N and P concentrations of the homogenized LSM samples were determined by acid digestion modified from Isaac and Johnson (1976). Briefly, 2 mL of LSM were mixed with 1.5 mL of a H_2SO_4–SeO_3 solution and 2 mL of H_2O_2 (30% v/v) in a 100-mL digester tube and allowed to react on the bench until foaming ceased. The tubes were then placed on a block digester set at 100°C for 30 min to evaporate water. After water evaporated, the block digester was set at 400°C, and the tubes were allowed to react for 40 min once the digester set point was reached. The tubes were cooled to room temperature, and the volume was completed to 100 mL with deionized water. The concentrations of NH_4^+ and PO_4^{3-} in the acid digests were measured with an automated continuous-flow injection analyzer (Model QuickChem 8000 FIA+; Lachat Instruments, Loveland, CO).

The mineral N content in LSM was determined by shaking 10 mL of LSM with 50 mL of 1 M KCl solution for 60 min. The extract was filtered with pre-washed (1 M KCl) filter papers (Whatman #42). Blank samples were used to detect any N contaminations. The NH_4^+, NO_3^-, and NO_2^- concentrations were measured in the extracts with the automated continuous-flow injection analyzer described previously. The sum of NO_3^-–N and NO_2^-–N contents accounted for <0.1% of total LSM mineral N.

Table 1. Actual amounts of N applied as mineral fertilizer (MF) and raw and treated liquid swine manure (LSM) in 2001, 2002, and 2003.

N source	2001	2002	2003	Average (2001–2003)
		— kg N ha⁻¹ —		
MF	140.0	140.0	140.0	140.0
LSM type				
Raw	145.3	142.1	150.0	145.8
Decanted	141.9	140.3	143.6	141.9
Filtered	148.7	140.1	143.4	144.1
Digested	137.6	139.3	144.7	140.5
Flocculated	142.3	140.0	140.4	140.9

Study Sites and Experimental Set-up

The experimental sites were located near Québec City (46°48′ N; 71°23′ W), Canada, on a loam (loamy, mixed, frigid, Typic Humaquept) and a sandy loam (sandy, mixed, frigid, Typic Dystrochrept), with mean annual temperature of 4°C and mean annual precipitation of 1100 mm. Before the experiment, the loam had been cropped to a pure stand of timothy, and the sandy loam had been cropped to a mix of timothy and red clover (*Trifolium pratense* L.). Both soils were plowed under in autumn 1999, and timothy (cv. Champ) was sown in spring 2000. At the beginning of the experiment in April 2001, the loam had a pH of 6.1, a total C content of 1.8 mg kg⁻¹, a total N content of 0.14 mg kg⁻¹, a clay content of 227 g kg⁻¹, and a sand content of 355 g kg⁻¹. The sandy loam had a pH of 6.5, a total C content of 2.1 mg kg⁻¹, a total N content of 0.14 mg kg⁻¹, a clay content of 171 g kg⁻¹, and a sand content of 726 g kg⁻¹.

Agronomy plots were established on both soils on April 2001 and arranged as a randomized complete block design with four replications. Each plot was 3 × 7 m. Seven treatments were applied on May 2001: unfertilized control; mineral fertilizer; and the raw, decanted, filtered, digested, and flocculated LSM. The various LSM were applied as evenly as possible on the surface of the plots with watering cans. There were 28 agronomy plots (seven treatments, four replications) at each experimental site. In addition to the 28 agronomy plots, six 0.5 × 2 m plots were established at each site to accommodate six wind tunnels to measure NH_3 emissions (Rochette et al., 2001). These small NH_3 plots were located about 2 m away from the agronomy plots and managed in the same way. Only one wind tunnel was used for each of the five LSM types and the mineral fertilizer at each site.

The experimental treatments were applied to the same plots for three consecutive years (2001, 2002, and 2003) and were applied to agronomy and NH_3 plots on the same day. The total amount of N applied for each growing season was set to ∼140 kg ha⁻¹ and split 80 kg ha⁻¹ in spring and 60 kg ha⁻¹ after the first harvest (recommended rates). The amounts of N applied in the various treatments are reported in Table 1. Spring applications were performed between 8 and 23 May of each year, and the applications after the first harvest were performed between 18 and 25 June (i.e., 1 to 6 d after harvest). All LSM types were applied based on their total N content. Nitrogen was applied as ammonium nitrate in the mineral fertilizer treatment. Phosphorus (50 kg P_2O_5 ha⁻¹ as superphosphate) and potassium (80 kg

K_2O ha⁻¹ as KCl) were supplied each year to the mineral fertilizer plots, whereas the LSM-treated plots received P and K only from the manure applied. The control plots did not receive fertilizers.

Gaseous N Emissions

Ammonia-nitrogen volatilization losses were monitored for 8 d after every N application using wind tunnels (Rochette et al., 2001; wind speed under tunnel, 1.3 m s⁻¹). On each application date, the entire surface (0.5 × 2 m) of the NH_3 plots was treated with mineral fertilizer or one of the five LSM types using watering cans and covered with wind tunnels within 2 min of application. The concentration of NH_3 in the air entering and leaving the tunnels was measured by bubbling the air in acid traps (0.005 M H_3PO_4). The acid traps were replaced at pre-determined intervals, and the concentration in NH_4^+ was measured by colorimetry (Nkonge and Ballance, 1982). The amounts of NH_3 emitted during each sampling period were calculated as the difference in NH_3 concentration between the air leaving and entering the tunnel multiplied by the air flow rate through the tunnel (Rochette et al., 2001). Total NH_3 loss after N application was calculated by summing the amounts of NH_3 emitted during all sampling periods. Cumulative NH_3 losses for a given year were calculated as the sum of the two 8-d measurement periods of that year.

Soil N_2O fluxes were measured on the agronomy plots at approximately 1, 3, 7, 14, and 21 d after every N application using non-flowthrough, non–steady-state chambers (Rochette and Hutchinson, 2005). One Plexiglas frame (0.3025 m² and 0.14 m height) was inserted to a depth of 0.1 m in every agronomy plot and left undisturbed for the duration of the experiment. On every sampling date, the frame was tightly fitted to a vented and insulated Plexiglas chamber (0.3 m in height) covering the same area as the frame. Air samples (20 mL) were collected from the chamber headspace at 0, 10, 20, and 30 min after chamber deployment. Air samples were collected with a syringe through a rubber septum and immediately transferred in pre-evacuated 12-mL glass vials (Exetainers; Labco, High Wycombe, UK). Gas samples and quality controls were analyzed for N_2O concentration within 10 d using a gas chromatograph equipped with an electron capture detector (Rochette and Hutchinson, 2005). The soil-surface N_2O emission rates were calculated using the equation and nonlinear model proposed by Rochette and Hutchinson (2005). N_2O losses were estimated for 21 d after every N application by linearly interpolating emission rates between measurement dates. Cumulative N_2O losses for a given year were calculated as the sum of the two 21-d measurement periods of that year.

Soil Sampling and Analyses

Soils were sampled on the same days as N_2O flux measurements. In each agronomy plot, three soil cores (0–0.2 m) were manually collected with a Dutch auger (0.053-m diam.) and pooled to make one composite sample per plot. Soil samples were immediately brought to the laboratory in plastic bags, and large soil aggregates were manually broken to make homogeneous samples. Exchangeable NH_4^+ and NO_3^- were measured on the day of soil sampling. Briefly, 20 g of field-moist soil were extracted with 100

mL of 1 *M* KCl solution for 60 min on a reciprocal shaker. The extracts were centrifuged (10 min, 3000 × *g*), filtered using pre-washed filter papers (Whatman #42), and frozen (–20°C) until analyzed by colorimetry with an automated continuous-flow injection analyzer as described for mineral N determination in LSM.

Forage Sampling and Analyses

The first harvest was taken between 13 and 20 June of each year, and a second harvest was taken between 26 July and 1 August. Dry matter yield was determined by harvesting forage at a 5-cm height above the soil from a 0.9 × 7-m area in each plot using a self-propelled flail forage harvester (Carter MGF Co., Brookston, IN). The harvested forage was weighed, and a fresh sample of approximately 500 g was taken from each plot, weighed, and dried at 55°C in a forced-draft oven for 3 d for determination of percentage DM. Dried samples were ground using a Wiley mill (Standars model 3; Arthur H. Thomas Co., Philadelphia, PA) fitted with a 1-mm screen. Forage N concentration was determined using a method adapted from Isaac and Johnson (1976) as described for total N determination in LSM. Crop N uptake was calculated as the product of forage N concentration and DM yield.

Statistical Analyses

Analyses of variance were performed using the GLM procedure of SAS (SAS Insitute, 1999) to test the treatment effects on cumulative NH_3 and N_2O losses, and forage DM yield and N uptake. When the treatment effect was significant ($P < 0.05$), a LSD test was performed to determine significant differences among treatments. The ANOVA were performed separately for each experimental site for each experimental year (2001, 2002, and 2003) and for the average values obtained over the 3 yr. Because wind tunnels were not replicated at each experimental site, we could not analyze the treatment effect on NH_3 loss at each site. We therefore considered sites as random effects, and the site × treatment interaction was used as the error term to test the treatment effect.

Results and Discussion

Liquid Swine Manure Characteristics

The composition of each LSM type was consistent across years, with CVs lower than 20% for most characteristics measured in the manure collected from 2001 to 2003 (Table 2). Solids removal increased the pH of LSM, particularly with anaerobic digestion, which increased pH by almost one unit. This effect of anaerobic digestion was noted in previous studies (Pain et al., 1990; Kirchmann and Lundvall, 1993; Chantigny et al., 2004) and was mainly attributed to the decomposition of volatile fatty acids into methane during the digestion process (Sommer and Husted, 1995).

Solids removal by natural decantation, filtration, and anaerobic digestion induced a marked reduction in LSM DM and total C content but had a lesser impact on N content (Table 2). Natural decantation removed 46% of the initial LSM P, whereas filtration removed 28% of P, indicating that this nutrient was associated with the solids removed by mechanical separation. More DM and P remained in the filtered than in the decanted LSM, likely because

Table 2. Selected properties of the different types of liquid swine manure (LSM).

Properties	LSM type				
	Raw	Decanted	Filtered	Digested	Flocculated
pH	7.4 (0.2) †	7.6 (0.2)	7.7 (0.2)	8.3 (0.1)	7.7 (0.1)
	——————————— kg m⁻³ ———————————				
Dry matter	45.8 (4.4)	25.9 (3.1)	30.7 (2.1)	16.5 (2.8)	13.0 (1.7)
Total C	24.3 (1.7)	18.2 (1.2)	19.9 (2.3)	8.8 (1.4)	8.1 (0.3)
Nitrogen					
Total	5.46 (0.10)	5.09 (0.23)	4.93 (0.32)	4.84 (0.02)	3.04 (0.36)
TAN‡	3.76 (0.20)	3.52 (0.29)	3.15 (0.28)	3.37 (0.25)	2.30 (0.06)
Total P	1.14 (0.16)	0.61 (0.18)	0.82 (0.12)	0.82 (0.08)	0.18 (0.02)

† Average values over 3 yr (2001, 2002, and 2003). Numbers in parentheses are SDs among years.

‡ TAN, total ammoniacal N.

more of the fine fraction of LSM solids and the associated P were able to pass through the filtration media. Anaerobic digestion has little effect on P content of LSM because solids removal is mediated through the biological conversion of organic C into methane (Massé et al., 1996). The decrease in the ammoniacal N and P contents of LSM after anaerobic digestion could be mainly attributed to the precipitation of struvite ($NH_4MgPO_4 \cdot 6H_2O$), which readily forms in anaerobic digesters under alkaline conditions (Sommer and Husted, 1995; Shu et al., 2006). The struvite precipitates at the bottom and on the side walls of the anaerobic digester and was not present in the liquid fraction used in the present study.

The flocculated LSM originated from a different LSM source for which we had no sample of the raw manure. Therefore, it cannot be directly compared with the other LSM types. However, both hog operations were managed in similar ways in terms of animal diet, and both LSM sources used for treatment were collected as relatively fresh material from the transfer tanks. In addition, LSM from finishing hog operations in the province of Quebec contains on average 35 kg DM m⁻³, 3.5 kg N m⁻³, and 0.9 kg P m⁻³ (CRAAQ, 2003). Assuming that the original LSM used for flocculation was similar to the provincial average, we estimated that the flocculation process removed about 75 to 80% of initial P from LSM.

Overall, the various treatments used to remove solids reduced LSM DM content with a consequent reduction in P but a minimal effect on N. Such effects of solids removal were previously reported for various methods of mechanical separation (Pain et al., 1978; Møller et al., 2000; Sørensen and Thomsen, 2005). In the present study, we found that chemical separation (flocculation) and anaerobic digestion can also reduce DM and P content of LSM (Table 2). As a result, the N/P ratio of LSM increased after solids removal, especially for the flocculated LSM, which had a N/P ratio of 17.

Soil Mineral N Content

Because mineral fertilization was applied as NH_4NO_3 and 60 to 80% of LSM N was present as NH_4^+ at time of application (Table 2), soil NH_4^+ content was markedly increased in both soils after every N application (Fig. 1, 2). This increase was generally similar among N treatments but was short-lived as the soil NH_4^+ content generally returned to values similar to the unfertilized
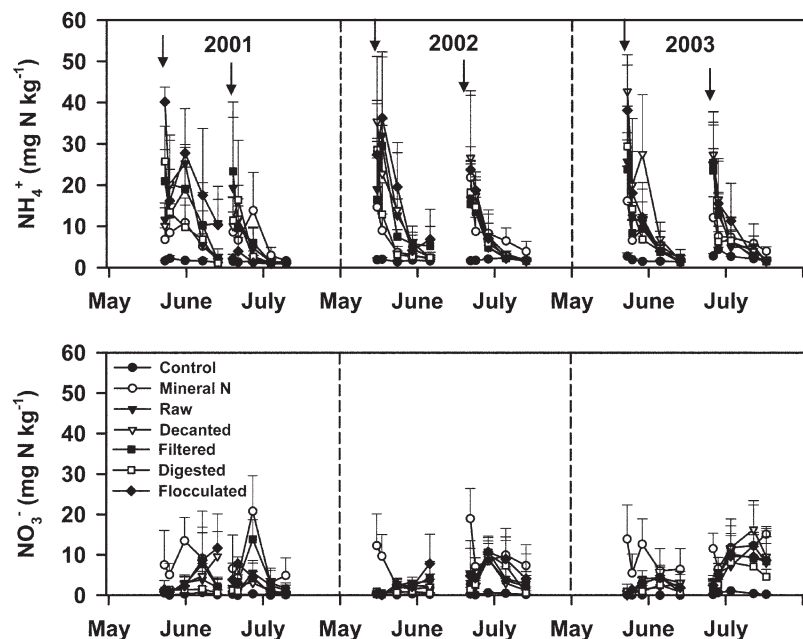
Fig. 1. Soil NH_4^+ and NO_3^- concentrations in the 0- to 0.2-m soil layer after application of mineral fertilizer and various types of liquid swine manure to a loam in 2001, 2002, and 2003. Each year, N sources were applied in spring and after the first harvest of hay. Arrows indicate the time of N applications. Vertical bars indicate SD of the means.

control 7 to 21 d after application. Soil NO_3^- content increased immediately after application of NH_4NO_3 but increased gradually for 1 to 2 wk in manured plots as a result of nitrification of NH_4^+ applied with LSM. The increase in soil NO_3^- content did not compensate for the decrease in soil NH_4^+ content, indicating that

Fig. 2. Soil NH_4^+ and NO_3^- concentrations in the 0- to 0.2-m soil layer after application of mineral fertilizer and various types of liquid swine manure to a sandy loam in 2001, 2002, and 2003. Each year, N sources were applied in spring and after the first harvest of hay. Arrows indicate the time of N applications. Vertical bars indicate SD of the means.

amounts of NO_3^- in soil samples were the net result between nitrification and other NO_3^-–consuming processes. Some NO_3^- leaching below 20 cm depth might have occurred after precipitation, especially in the sandy loam (Chantigny et al., 2001), but this was not monitored in the present study. The rapid decrease in soil NH_4^+ content was most likely due to plant N uptake and microbial immobilization, which are typically high under forage grass systems (Schimel, 1986; Recous et al., 1997). As a result, soil NO_3^- content generally remained below 15 mg N kg^{-1}, and soil mineral N content ($NH_4^+ + NO_3^-$) was generally close to background levels 21 d after N application (Fig. 1, 2).

Ammonia Volatilization

Overall, the cumulative NH_3 losses were two to six times greater with the various LSM types than with the mineral fertilizer (Table 3). This is in agreement with high emissions generally reported after surface application of alkaline, NH_4^+–rich manures to agricultural soils (Génermont, 1996; Sommer and Hutchings, 2001). When averaged across years, NH_3 losses were significantly lower with all the treated LSM types than with the raw LSM (Table 3). Lower NH_3 losses with treated LSM can be partly explained by their lower solids content (Table 2), which increased the infiltration rate of LSM in soil, thereby reducing the period of time during which LSM NH_4^+ was exposed to the atmosphere (Pain et al., 1990; Stevens et al., 1992; Wulf et al., 2002a).

However, when averaged across years, filtered LSM emitted significantly less NH_3 than the decanted and digested LSM (Table 3) despite a higher DM content (Table 2). Greater NH_3 emissions with digested than with filtered LSM could be explained by the higher pH of the digested LSM, which increases its potential for NH_3 volatilization (Pain et al., 1990; Wulf et al., 2002a). This explanation does not hold to explain the difference in NH_3 emissions between the decanted and filtered LSM because their pH was similar (Table 2). It is possible that the solids remaining in the filtered LSM were smaller in size than solids in the decanted LSM, which could result in a faster infiltration of the former. However, a detailed investigation of the physical and chemical nature of LSM solids was beyond the scope of this study. Averaged across years, the proportions of applied N emitted as NH_3 in the present study were 4% for ammonium nitrate, 15% for raw LSM, and 10 to 13% for the various treated LSM.

Despite similar N application rates among years, the amounts of NH_3 volatilized in 2003 were 1.6 to 4.1 times higher than in 2001 and 2002 for all LSM types (Table 3). Inter-annual variations in NH_3 volatilization were likely the
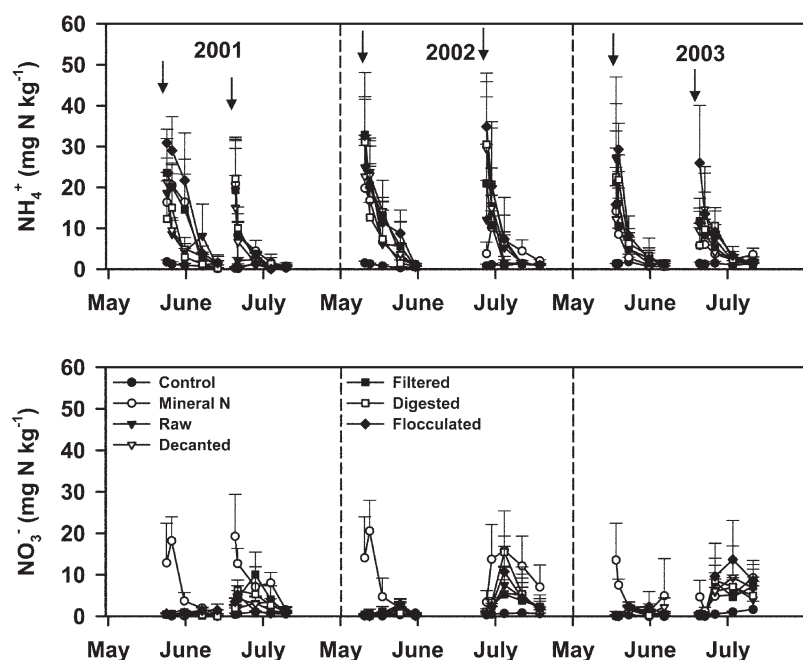
result of variations in air temperature and soil water content because manure characteristics and sward height at the time of LSM application (1–6 d after harvest) were similar among years. Indeed, high soil water content in 2003 (Table 4) likely impeded the infiltration of LSM, thereby favoring NH_3 volatilization (Génermont, 1996; Sommer and Hutchings, 2001). Soil water content was also high at the time of spring application in 2002 (Table 4). However, we hypothesize that the cooler air temperature (6–10°C lower than in spring 2001 and 2003) counterbalanced the stimulating effect of higher soil water content on NH_3 volatilization.

This effect of soil water content on NH_3 emissions was particularly strong for the digested LSM, which was associated with emissions 4.1 times higher in 2003 than in 2001 (Table 3). The more alkaline pH of the digested LSM increased its potential for NH_3 loss as compared with the other LSM types (Pain et al., 1990; Wulf et al., 2002a; Chantigny et al., 2004). The NH_3 loss from digested LSM was therefore more responsive than the other LSM types to variations in soil water content at time of application.

Nitrous Oxide Emissions

In the present study, N_2O was assumed to derive mostly from denitrification because significant N_2O emissions were essentially and consistently measured after precipitation (data not shown). The estimates of N_2O losses were 2.6 to 5.1 times lower in the sandy loam than in the loam soil (Table 5). This is likely because of the coarser texture and better drainage/aeration conditions in the sandy loam, which were less conducive to denitrification than in the loam.

The N treatments generally induced greater N_2O losses than the unfertilized control in both soils (Table 5), reflecting the stimulating effect of N addition to N_2O-producing processes. In addition, the raw, decanted, filtered, and flocculated LSM generally resulted in larger N_2O emissions than the mineral fertilizer. Higher N_2O emissions with liquid manure than mineral fertilizer were reported under field conditions (Petersen, 1999; Rochette et al., 2000; van Groenigen et al., 2004). It is likely that, in addition to the N applied, the easily decomposable C present in LSM further stimulated N_2O production by denitrifiers (Paul and Beauchamp, 1989; Rochette et al., 2000). The N_2O losses were similar among the raw, decanted, filtered, and flocculated LSM (Table 5), suggesting that even the lower C content in the flocculated LSM was sufficient to substantiate the denitrifying population.

The digested LSM was generally associated with significantly lower N_2O losses than the other LSM types (Table 5). Losses were 54 to 69% lower with the digested than with raw LSM in the loam soil and 17 to 71% lower in the sandy loam. This apparent depressive effect on soil N_2O emissions was also reported by others for digested cattle manure (Rubaek et al., 1996; Petersen, 1999; Amon et al., 2006) and swine manure (Vallejo et al., 2006). Nyberg et al. (2004) concluded that some compounds present in anaerobically digested manure may have a depressive effect on soil ammonia oxidizers, thereby reducing the supply of substrate for N_2O production through nitrification and denitrification. Alternatively, Vallejo et al. (2006) argued that because most easily degradable C present in manure is decomposed during anaerobic digestion, the

Table 3. Cumulative NH_3 losses measured† after application of mineral fertilizer (MF) and raw and treated liquid swine manure (LSM) to timothy in 2001, 2002, and 2003 (values are means of the two soil types).

N source	2001	2002	2003	Mean (2001–2003)
	kg NH_3–N ha^{-1}			
MF	ND	3.4	8.3	5.9
LSM type				
Raw	13.5	19.6	32.7	22.0
Decanted	13.2	13.4	26.7	17.8
Filtered	8.6	12.9	20.6	14.0
Digested	7.5	15.1	30.7	17.7
Flocculated	12.6	13.7	23.3	16.5
LSD‡	NS	6.9	15.1	3.3

† NH_3 emissions were measured for 8 d after each of the spring and summer applications and summed to estimate the cumulative NH_3–N loss for the growing season. ND, not determined.

‡ Wind tunnels were not replicated at each experimental site, and the AVOVA for NH_3 losses was performed with sites considered as replications. LSD values are presented only when $P < 0.05$. NS, not significant.

C remaining in the digested manure is more stable and, therefore, less likely to stimulate denitrification and N_2O production as compared with the undigested manure.

Crop Yield and N Uptake

Application of mineral fertilizer and LSM significantly increased forage DM yield and N uptake as compared with the unfertilized control (Tables 6 and 7), except for the loam soil in 2002. Presumably, environmental conditions were particularly favorable to indigenous soil N mineralization in the loam soil in 2002, which resulted in high crop N uptake and DM yield in all experimental treatments including the unfertilized control.

Forage DM yields were significantly higher for the mineral fertilizer than for most LSM types in 2001 in both soils but were similar for most N treatments in 2002 and 2003 (Table 6). When averaged over the 3 yr of experiment, most LSM types produced DM yields similar to the mineral fertilizer in the loam soil, whereas the digested LSM consistently produced forage DM yields similar to the mineral fertilizer across years and sites. Despite some statistically significant differences, forage DM yields obtained with the various LSM types were on average 90 to 99% of those obtained with the mineral fertilizer. Because of its high mineral N content, LSM was found to be an efficient fertilizer to produce forage grasses (Burns et al., 1987, 1990; Adeli et al., 2005). Moreover, Adeli

Table 4. Soil water content and air temperature† at time of N application to a loam and a sandy loam cropped to timothy in 2001, 2002, and 2003.

	Loam				Sandy loam			
	Soil water content		Air temperature		Soil water content		Air temperature	
Year	May	June	May	June	May	June	May	June
	g kg^{-1}		°C		g kg^{-1}		°C	
2001	154	162	15.8	19.1	128	217	16.0	18.4
2002	227	213	6.0	18.3	220	207	6.5	20.2
2003	295	255	12.7	20.6	278	247	12.2	17.4

† Values are average for the first 3 d after each N application.

Table 5. Cumulative N$_2$O–N losses measured† after the application of mineral fertilizer (MF) and raw and treated liquid swine manure (LSM) to a loam and a sandy loam cropped to timothy in 2001, 2002, and 2003.

N source	Loam				Sandy loam			
	2001	2002	2003	Mean (2001–2003)	2001	2002	2003	Mean (2001–2003)
	g N$_2$O–N ha^{-1}							
Control	357	84	101	181	72	64	40	59
MF	1219	629	1063	970	342	295	93	243
LSM type								
Raw	1691	1056	1728	1491	266	720	256	414
Decanted	1256	1319	902	1159	343	785	187	439
Filtered	1595	1486	2168	1750	274	523	241	346
Digested	666	481	530	559	182	206	213	200
Flocculated	1638	1135	1132	1302	255	445	92	264
LSD‡	747	493	1014	376	128	442	128	205

† N$_2$O emissions were measured for 21 d after each of the spring and summer applications (total of 42 d) and interpolated to estimate the cumulative N$_2$O–N loss for the growing season.

‡ LSD values are presented only when $P < 0.05$.

et al. (2005) reported similar forage DM yield and quality with LSM and ammonium nitrate.

In the present study, crop N uptake was most of the time significantly lower with the various LSM types than with the mineral fertilizer (Table 7), indicating that manure-derived N was less available to plant. This difference might be due to the presence of organic N in LSM (Table 2), which is not readily available to the plant. However, the difference may also be explained by the significantly greater NH$_3$ volatilization with LSM than with mineral fertilizer (Table 3) because NH$_3$ volatilization directly decreased the amounts of readily available manure N.

Even though the differences were not always statistically significant, crop N uptake tended to be higher with treated than with raw LSM (Table 7). Increased plant N recovery after solids removal was reported by Rubaek et al. (1996), Mattila et al. (2003), and Morris and Lathwell (2004) for cattle manure and by Sørensen and Thomsen (2005) for swine manure. On average across sites and LSM types, LSM treatments reduced NH$_3$ losses by 5.5 kg N ha^{-1} (Table 3), whereas crop uptake was increased by 4.5 kg N ha^{-1} in the loam and 12.8 kg N ha^{-1} in the sandy loam (Table 7). This is in agreement with Rubaek et al. (1996), who reported increased plant N uptake only where treated manure

resulted in lower NH$_3$ losses than untreated manure. We therefore hypothesized that the greater crop N uptake with the treated than with the raw LSM was partly due to the reduction in NH$_3$ losses. However, the lower C content of the treated LSM may also have resulted in a greater N availability compared with raw LSM because the decomposition of LSM C may induce soil N immobilization (Kirchmann and Lundvall, 1993), thereby interfering with plant N uptake.

Perspective and Conclusions

The present study looked at gaseous N emissions and plant N uptake from only the liquid fraction of treated hog manure even though a DM-rich fraction is often produced at the same time. The main purpose for hog manure treatment in the area of study is to extract and export P from operations facing surplus P problems due to local regulations for field application of manure based on crop P requirements. In this situation, only the liquid fraction remains on-farm, whereas the DM-rich fraction is exported to composting or organic fertilizer plants. In other areas, however, manure treatment can be viewed as an option for optimized on-farm nutrient management (Møller et al., 2000). In this case, a thorough assessment of the liquid and DM-rich fractions is required because some benefits found with the liquid fraction may be partly offset by lower N efficiency and greater environmental losses with the DM-rich fraction (Sørensen and Thomsen, 2005; Amon et al., 2006).

Removing solids in LSM by mechanical, chemical, or biological means seemed to be a valid option to reduce gaseous N emissions after land application. All LSM treatments reduced NH$_3$ emissions (average reduction of 25%) relative to the untreated LSM, and anaerobic digestion also significantly reduced N$_2$O emissions.

Liquid swine manure, treated or untreated, offers an alternative to mineral fertilizers for the production of forage grasses under the cool and humid climate of eastern Canada. Forage DM yields with LSM were on average >90% of those obtained with mineral fertilizer. However, management practices that minimize NH$_3$ volatilization should be used where LSM is applied because it emits significantly more NH$_3$ than mineral fertilizers, such as ammonium nitrate. In perennial forage systems where incorporation of manure may disturb the rooting system, removing solids from LSM before land application is an option to significantly reduce NH$_3$ losses and thereby to improve the fertilizer efficiency of LSM.

Table 6. Seasonal forage dry matter yield of timothy harvested twice a year on a loam and a sandy loam after the application of mineral fertilizer (MF) and raw and treated liquid swine manure (LSM) in 2001, 2002, and 2003.

N source	Loam				Sandy loam			
	2001	2002	2003	Mean (2001–2003)	2001	2002	2003	Mean (2001–2003)
	Mg DM ha^{-1}							
Control	3.66	7.51	3.94	5.04	5.63	4.27	2.89	4.26
MF	6.11	7.43	5.71	6.42	9.06	8.52	5.25	7.61
LSM type								
Raw	5.12	7.97	5.32	6.14	7.93	7.45	5.29	6.89
Decanted	5.14	7.40	5.39	5.98	8.09	8.04	5.33	7.15
Filtered	5.70	7.65	5.46	6.27	8.15	7.51	4.84	6.83
Digested	5.93	7.37	5.71	6.34	8.78	8.40	5.19	7.46
Flocculated	5.19	7.87	5.25	6.10	7.51	8.65	5.07	7.08
LSD†	0.35	NS	0.39	0.40	0.49	0.54	0.50	0.37

† LSD values are presented only when $P < 0.05$. NS, not significant.

Table 7. Seasonal crop N uptake of timothy harvested twice a year on a loam and a sandy loam after the application of mineral fertilizer (MF) and raw and treated liquid swine manure (LSM) in 2001, 2002, and 2003.

	Loam				Sandy loam			
N source	2001	2002	2003	Mean (2001–2003)	2001	2002	2003	Mean (2001–2003)
	kg N ha⁻¹							
Control	68	175	63	102	80	66	44	64
MF	151	202	124	159	192	178	109	160
LSM type								
Raw	112	199	111	141	133	140	93	122
Decanted	118	195	113	142	152	157	101	136
Filtered	128	195	106	143	153	142	88	128
Digested	129	196	124	149	161	160	97	139
Flocculated	123	206	113	148	147	166	94	136
LSD†	9	NS	11	10	10	12	8	8

† LSD values are presented only when $P < 0.05$. NS, not significant.

References

Adeli, A., J.J. Varco, K.R. Sistani, and D.E. Rowe. 2005. Effects of swine lagoon effluent relative to commercial fertilizer applications on warm-season forage nutritive value. Agron. J. 97:408–417.

Amon, B., V. Kryvoruchko, T. Amon, and S. Zechmeister-Boltenstern. 2006. Methane, nitrous oxide and ammonia emissions during storage and after application of dairy cattle slurry and influence of slurry treatment. Agric. Ecosyst. Environ. 112:153–162.

Burns, J.C., L.D. King, and P.W. Westerman. 1990. Long-term swine lagoon effluent applications on coastal bermudagrass: I. Yield, quality, and element removal. J. Environ. Qual. 19:749–756.

Burns, J.C., P.W. Westerman, L.D. King, M.R. Overcash, and G.A. Cummings. 1987. Swine manure and lagoon effluent applied to a temperate forage mixture: I. Persistence, yield, quality, and elemental removal. J. Environ. Qual. 16:99–105.

Chantigny, M.H., P. Rochette, and D.A. Angers. 2001. Short-term C and N dynamics in a soil amended with pig slurry and barley straw: A field experiment. Can. J. Soil Sci. 81:131–137.

Chantigny, M.H., P. Rochette, D.A. Angers, D. Massé, and D. Côté. 2004. Ammonia volatilization and selected soil characteristics following application of anaerobically digested pig slurry. Soil Sci. Soc. Am. J. 68:306–312.

Clemens, J., R. Vandré, M. Kaupenjohann, and H. Goldbach. 1997. Ammonia and nitrous oxide emissions after landspreading of slurry as influenced by application technique and dry matter-reduction: II. Short term nitrous oxide emissions. Z. Pflanzenernähr. Bodenk. 160:491–496.

CRAAQ. 2003. Charges fertilisantes des effluents d'élevage: période transitoire. Valeurs références–Production porcine. Centre de Références en Agriculture et Agroalimentaire du Québec. Québec City, Canada. Available at http://pub.craaq.qc.ca/Transit/9-Porcine.pdf (verified 18 June 2007).

Génermont, S. 1996. Modélisation de la volatilisation d'ammoniac après épandage de lisier sur parcelle agricole. Ph.D. thesis, Université Paul Sabatier, Laboratoire d'Aérologie, Toulouse, France.

Isaac, R.A., and W.C. Johnson. 1976. Determination of total nitrogen in plant tissue, using a block digestor. J. Assoc. Off. Anal. Chem. 59:98–100.

Kirchmann, H., and A. Lundvall. 1993. Relationship between N immobilization and volatile fatty acids in soil after application of pig and cattle slurry. Biol. Fertil. Soils 15:161–164.

Massé, D.I., N.K. Patni, R.L. Droste, and K.J. Kennedy. 1996. Operation strategies for psychrophilic anaerobic digestion of swine manure slurry in sequencing batch reactors. Can. J. Civ. Eng. 23:1285–1294.

Mattila, P.K., and E. Joki-Tokola. 2003. Effect of treatment and application technique of cattle slurry on its utilization by ley: I. Slurry properties and ammonia volatilization. Nutr. Cycling Agroecosyst. 65:221–230.

Mattila, P.K., E. Joki-Tokola, and R. Tanni. 2003. Effect of treatment and application technique of cattle slurry on its utilization by ley: II. Recovery of nitrogen and composition of herbage yield. Nutr. Cycling Agroecosyst. 65:231–242.

Møller, H.B., I. Lund, and S.G. Sommer. 2000. Solid-liquid separation of livestock slurry: Efficiency and cost. Bioresour. Technol. 74:223–229.

Morris, D.R., and D.J. Lathwell. 2004. Anaerobic digested dairy manure as fertilizer for maize in acid and alkaline soils. Commun. Soil Sci. Plant Anal. 35:1757–1771.

Nkonge, C., and G.M. Ballance. 1982. A sensitive colorimetric procedure for nitrogen determination in micro-Kjeldahl. J. Agric. Food Chem. 30:416–420.

Nyberg, K., I. Sundh, M. Johansson, and A. Schnürer. 2004. Presence of potential ammonia oxidation (PAO) inhibiting substances in anaerobic digestion residues. Appl. Soil Ecol. 26:107–112.

Pain, B.F., R.Q. Hepherd, and R.J. Pittman. 1978. Factors affecting the performances of four slurry separating machines. J. Agric. Eng. Res. 23:231–242.

Pain, B.F., T.H. Misselbrook, C.R. Clarkson, and Y.J. Rees. 1990. Odour and ammonia emissions following the spreading of anaerobically-digested pig slurry on grassland. Biol. Wastes 34:259–267.

Paul, J.W., and E.G. Beauchamp. 1989. Effect of carbon constituents in manure on denitrification in soil. Can. J. Soil Sci. 69:49–61.

Petersen, S.O. 1999. Nitrous oxide emissions from manure and inorganic fertilizers applied to spring barley. J. Environ. Qual. 28:1610–1618.

Recous, S., P. Loiseau, J.M. Machet, and B. Mary. 1997. Transformations et devenir de l'azote de l'engrais sous cultures annuelles et sous prairies. p. 105–120. In G. Lemaire and B. Nicolardot (ed.) Maîtrise de l'azote dans les agrosystèmes. Les Colloques, no. 83, INRA Editions, Paris, France.

Rochette, P., M.H. Chantigny, D.A. Angers, N. Bertrand, and D. Côté. 2001. Ammonia volatilization and soil nitrogen dynamics following fall application of pig slurry on canola crop residues. Can. J. Soil Sci. 81:515–523.

Rochette, P., and G.L. Hutchinson. 2005. Measuring soil respiration using chamber techniques. p. 227–266. In J. Hatfield and J.M. Baker (ed.) Micromet studies of the soil-plant-atmosphere continuum. Agron. Monogr. No. 47. ASA, CSSA, and SSSA, Madison, WI.

Rochette, P., E. van Bochove, D. Prévost, D.A. Angers, D. Côté, and N. Bertrand. 2000. Soil carbon and nitrogen dynamics following application of pig slurry for the 19th consecutive year: II. Nitrous oxide fluxes and mineral nitrogen. Soil Sci. Soc. Am. J. 64:1396–1403.

Rubaek, G.H., K. Henriksen, J. Petersen, B. Rasmussen, and S.G. Sommer. 1996. Effects of application technique and anaerobic digestion on gaseous nitrogen loss from animal slurry applied to ryegrass (Lolium perenne). J. Agric. Sci. (Cambridge) 126:481–492.

SAS Institute. 1999. SAS/STAT user's guide. Version 8. SAS Inst. Inc., Cary, NC.

Schimel, D.S. 1986. Carbon and nitrogen turnover in adjacent grassland and cropland ecosystems. Biogeochemistry 2:345–357.

Shu, L., P. Schneider, V. Jegatheesan, and J. Johnson. 2006. An economic

evaluation of phosphorus recovery as struvite from digester supernatant. Bioresour. Technol. 97:2211–2216.

Sommer, S.G., and S. Husted. 1995. The chemical buffer system in raw and digested animal slurry. J. Agric. Sci. (Cambridge) 124:45–53.

Sommer, S.G., and N.J. Hutchings. 2001. Ammonia emission from field applied manure and its reduction. Eur. J. Agron. 15:1–15.

Sørensen, P., and I.K. Thomsen. 2005. Separation of pig slurry and plant utilization and loss of nitrogen-15-labeled slurry nitrogen. Soil Sci. Soc. Am. J. 69:1644–1651.

Stevens, R.J., R.J. Laughlin, and J.P. Frost. 1992. Effects of separation, dilution, washing, and acidification on ammonia volatilization from surface-applied cattle slurry. J. Agric. Sci. (Cambridge) 119:383–389.

Thompson, R.B., B.F. Pain, and D.R. Lockyer. 1990. Ammonia volatilization from cattle slurry following surface application to grassland: I. Influence of mechanical separation, changes in chemical composition during volatilization, and the presence of the grass sward. Plant Soil 125:109–117.

Vallejo, A., U.M. Skiba, L. Garcia-Torres, A. Arce, S. Lopez-Fernandez, and L. Sanchez-Martin. 2006. Nitrogen oxides emission from soils bearing a potato crop as influenced by fertilization with treated pig slurries and composts. Soil Biol. Biochem. 38:2782–2793.

Vandré, R., J. Clemens, H. Goldbach, and M. Kaupenjohann. 1997. NH_3 and N_2O emissions after landspreading of slurry as influenced by application technique and dry matter- reduction: I. NH_3 emissions. Z. Pflanzenernähr. Bodenk. 160:303–307.

van Groenigen, J.W., G.J. Kasper, G.L. Velthof, A. van den Pol-van Dasselaar, and P.J. Kuikman. 2004. Nitrous oxide emissions from silage maize fields under different mineral nitrogen fertilizer and slurry applications. Plant Soil 263:101–111.

Wulf, S., M. Maeting, and J. Clemens. 2002a. Application technique and slurry co-fermentation effects on ammonia, nitrous oxide, and methane emissions after spreading: I. Ammonia volatilization. J. Environ. Qual. 31:1789–1794.

Wulf, S., M. Maeting, and J. Clemens. 2002b. Application technique and slurry co-fermentation effects on ammonia, nitrous oxide, and methane emissions after spreading: II. Greenhouse gas emissions. J. Environ. Qual. 31:1795–1801.