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Greenhouse Gas Emissions from Two Soils Receiving Nitrogen Fertilizer and Swine Manure Slurry

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The interactive effects of soil texture and type of N fertility (i.e., manure vs. commercial N fertilizer) on N_2O and CH_4 emissions have not been well established. This study was conducted to assess the impact of soil type and N fertility on greenhouse gas fluxes (N_2O, CH_4, and CO_2) from the soil surface. The soils used were a sandy loam (789 g kg^{-1} sand and 138 g kg^{-1} clay) and a clay soil (216 g kg^{-1} sand, and 415 g kg^{-1} clay). Chamber experiments were conducted using plastic buckets as the experimental units. The treatments applied to each soil type were: (i) control (no added N), (ii) urea-ammonium nitrate (UAN), and (iii) liquid swine manure slurry. Greenhouse gas fluxes were measured over 8 weeks. Within the UAN and swine manure treatments both N_2O and CH_4 emissions were greater in the sandy loam than in the clay soil. In the sandy loam soil N_2O emissions were significantly different among all N treatments, but in the clay soil only the manure treatment had significantly higher N_2O emissions. It is thought that the major differences between the two soils controlling both N_2O and CH_4 emissions were cation exchange capacity (CEC) and percent water-filled pore space (%WFPS). We speculate that the higher CEC in the clay soil reduced N availability through increased adsorption of NH_4^+ compared to the sandy loam soil. In addition the higher average %WFPS in the sandy loam may have favored higher denitrification and CH_4 production than in the clay soil.

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Total emissions of greenhouse gasses (GHGs) can be significantly altered by agricultural practices. Increases in N_2O flux are impacted by human activity (Rochette et al., 2000), and it has been estimated that agriculture contributes from 60 to 80% of total N_2O emissions on a global scale (Isermann, 1994; Janzen et al., 1998; Cameron et al., 2000; Dalal et al., 2003). A major factor influencing N_2O emissions from agricultural lands is N application (Mosier et al., 1982; Kaiser et al., 1998; Rochette et al., 2000; de Klein et al., 2001; Yamulki and Jarvis, 2002; Dalal et al., 2003). Nitrogen fertilization can significantly enhance N_2O emissions and it is generally recognized that as N inputs increase, N_2O emissions also increase (Mosier et al., 1982; Eichner, 1990). The relationship between N inputs and N_2O emissions is commonly described as linear (Bouwman, 1996; Gregorich et al., 2005; IPCC, 2006), although recently, McSwiney and Robertson (2005) present evidence that the relationship between N inputs and N_2O emissions may exhibit a threshold effect.

It is also recognized that, despite the common assumption of linearity between N inputs and N_2O emissions, soil systems are quite complex (Bouwman et al., 2002; Mosier and Parkin, 2007). The form of N fertilizer applied has been observed to influence N_2O emissions (Breitenbeck et al., 1980; Eichner, 1990) as well as N fertilizer placement in the soil (Bremner et al., 1981). Land application of animal waste also increases N_2O emission (Mosier et al., 1998; Petersen, 1999). According to Kaiser and Ruser (2000), from 0.74 to 2.86% of slurry N was emitted as a N_2O annually, whereas de Klein et al. (2001) reported annual N-N_2O losses of manure N ranging from 0 to 5%. The largest emission of N_2O in the range of 100 to 150 g N_2O ha^{-1} d^{-1} occurred within 1 to 2 d after the injection of slurry (Comfort et al., 1990).

Nitrogen fertilization also reduces soil CH_4 oxidation (Steudler et al., 1989; Schimel and Gulledge, 1998). Soils can be a source or a sink for CH_4 depending on soil type, aeration, and nitrogen availability (Chan and Parkin, 2001b; Le Mer and Roger, 2001). In upland soils CH_4 oxidation is typically greater than methano-

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Abbreviations: CEC, cation exchange capacity; GHG, greenhouse gas; IPCC, Intergovernmental Panel on Climate Change; SOC, soil organic carbon; UAN, urea ammonium nitrate; %WFPS, percent water-filled pore space.

genesis (Conrad, 1996; Hütsch, 2001). The potential for different ecosystems to serve as a sink for atmospheric CH_4 varies from 1 to 2 kg CH_4–C ha^{-1} yr^{-1}, however, different sources of N inputs can considerably suppress CH_4 oxidation rates (Smith et al., 2000; Suwanwaree and Robertson, 2005).

Pork production is a major agricultural enterprise in the Midwest and results in the production of large quantities of liquid or semi-liquid manure slurry. Direct injection of this swine manure has become a common technique for land application (Hatfield et al., 1998). The greater contact of injected slurry with soil can induce favorable conditions for N_2O and CH_4 formation probably because of the restricted aeration at the injected slurry treatment (Flessa and Besse, 2000; Wulf et al., 2002). However, Dendooven et al. (1998) did not find differences in N_2O and CH_4 emission between injected and surface-applied pig slurry from loamy soil.

Development of a comprehensive understanding of the complexities of the interactions of soil/environmental/and management factors and their effects on the biology of N_2O production and CH_4 cycling is a daunting task. This work attempts to explore some of the interactions between soil type and N fertilization in controlled laboratory experiments using two different soils and three N fertility regimes. The specific objectives of this study were: (i) compare N_2O, CO_2, and CH_4 fluxes from soil receiving swine manure slurry and a commercial N source, and (ii) examine the interactive effects of soil type and N fertility treatment on N_2O and CH_4 emissions.

Materials and Methods

Experimental Design

The experiment was performed in a controlled environmental chamber programmed for a 14 h light period, 18°C day temperature, and 15°C night temperature. The treatments were organized in a randomized complete block design with 2 × 3 factorial arrangement of soil type and N source treatments. The soils were a sandy loam [classified as a Storden fine-loamy, mixed, mesic Typic Udorthents, (USDA, 1981)] and a clay [classified as a Webster fine-loamy, mixed, mesic Typic Haplaquolls, (USDA, 1981)]. The soils selected for the experiment were collected from Iowa State University Agriculture Experiment Station located in Boone County in central Iowa. Both soils had neutral pH although clayey soil had 2.5 times higher concentration of soil organic matter (SOC) and cation exchange capacity (CEC) than sandy soil (Table 1).

The N source treatments were: (i) control with no N source, (ii) urea-ammonium nitrate (UAN) fertilizer N, and (iii) swine manure slurry N. The treatments were replicated four times and the experiment was conducted for 8 wk. The fertilizer N used was a UAN solution (32% N with density 1.32 g cm^{-3}). The swine manure slurry was collected from manure storage tanks and contained 1.8 g kg^{-1} total N, 0.8 g kg^{-1} of ammonia N, and 24 g kg^{-1} dry matter. Both N fertilizers were applied at a rate of 200 kg N ha^{-1}. Details of N application are provided below.

Experiment Setup

Plastic buckets (0.28 m in diameter and 0.35 m in height) containing soil without vegetation were used as the experimental units. Each bucket was equipped with a drainage system consisting of a 48 mm diameter and 60 mm long ceramic cup placed at the bottom with an air entry value of 50 kPa. The end of the ceramic cup was sealed with a rubber stopper which had plastic tubing inserted through its center to connect later to a vacuum pump that maintained a vacuum of 9.8 kPa and pulled any water that was collected in the ceramic cup into a collection flask. The bottom of the bucket and the ceramic cup were covered by 5 kg of coarse sand on which 12 kg of air-dried soil was placed. At the first watering, the buckets with soil and sand were treated with 4000 mL of 0.005 mol L^{-1} $CaCl_2$ to prevent soil aggregate dispersion. Suction was applied to the ceramic candles, the soil surface in each bucket was covered with plastic, and the buckets were allowed to equilibrate for 24 h. After 24 h the buckets were weighed to determine water content at field capacity. At weekly intervals throughout the incubation, the buckets were weighed, the soil water content of each bucket was calculated, and water added to rewet the soil to field capacity. Percent water-filled pore space (%WFPS) was calculated from measurements of bulk density and soil water content. At field capacity the %WFPS for the sandy loam and clay soils were 54 and 49%, respectively.

Table 1. Properties of soil used in this experiment.

Properties	Sandy soil	Clayey soil
pH	6.9	7.0
Bulk density (Mg m^{-3})	1.34 (0.05)†	1.12 (0.02)
Sand (g kg^{-1} soil)	789 (12)	216 (7)
Silt (g kg^{-1} soil)	73 (19)	369 (6)
Clay (g kg^{-1} soil)	138 (13)	415 (13)
%WFPS at field capacity	54	48
Soil organic C (g kg^{-1} soil)	18.1 (0.8)	44.4 (1.6)
Dissolved organic C (mg kg^{-1} soil)	137 (5)	186 (14)
Total N (g kg^{-1} soil)	1.75 (0.10)	3.63 (0.20)
NO_3^- (mg N kg^{-1} soil)	13.5 (6.5)	26.7 (8.3)
NH_4^+ (mg N kg^{-1} soil)	0.11 (0.18)	0.13 (0.30)
CEC ($cmol_c$ kg^{-1} soil)	10.6 (0.4)	26.7 (0.7)

† Standard deviation of triplicate analyses.

Swine Manure Slurry and UAN

Manure slurry was applied by cutting a 5 cm wide × 5 cm deep trench into the soil surface, pouring slurry (684 mL) into the furrow, and covering it with the soil previously removed from the trench. A UAN solution (0.211 g N mL^{-1}) was injected at two points in each bucket (2.92 mL per point). There was a 15 cm separation between the two injection points. This method of application was chosen to simulate field application of UAN with a spoke injector. Additional water (680 mL) was surface applied to the UAN and control treatments to match the amount of liquid applied in the swine manure slurry.

Greenhouse Gas Emissions

Greenhouse gases flux measurements were performed every day over the first 2 wk and every second day over the remaining period by placing vented PVC chambers (30 cm diameter × 10 cm tall) on the buckets, and collecting gas samples at 0, 30, and 60 min following chamber deployment. Gas samples were taken by 10 mL polypropylene syringes and immediately injected into evacuated glass vials (6 mm) fit with butyl rubber stoppers. Gas

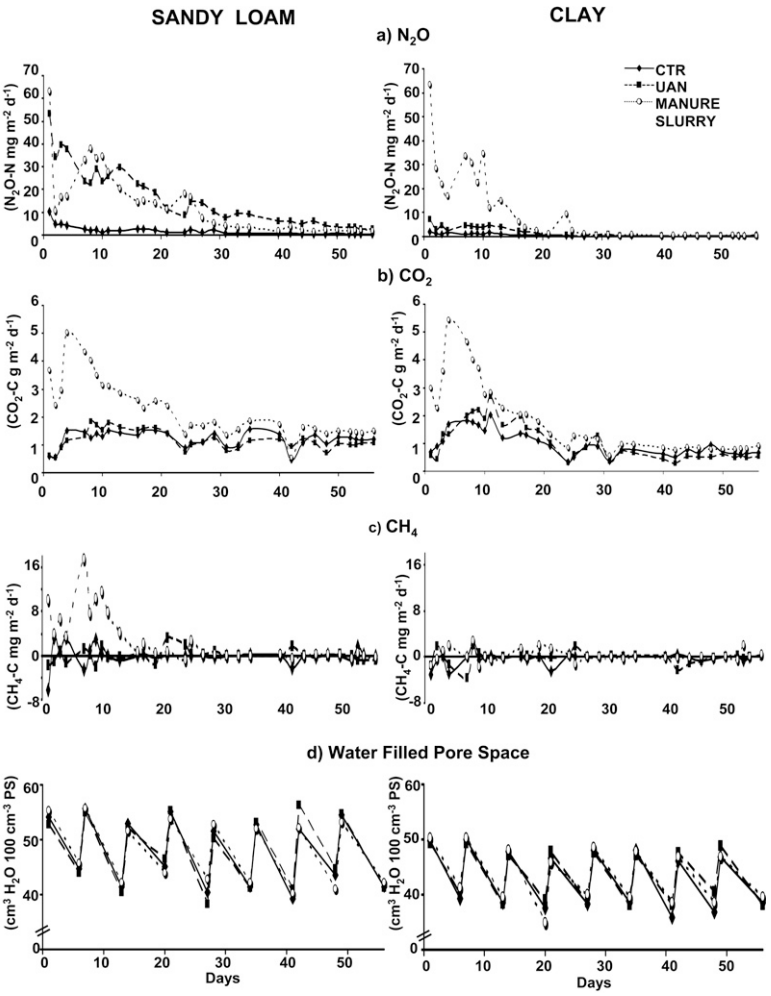

Fig. 1. Greenhouse gas emission and soil water-filled pore space over a 56-d period.

samples were analyzed for N_2O, CO_2, and CH_4 with an SRI gas chromatograph and introduced into the gas chromatograph using an auto sampler described by Arnold et al. (2001). Nitrous oxide was measured using a ^{63}Ni electron capture detector (ECD), with a stainless steel column (HaySepD, 0.3175 cm diameter × 74.54 cm long). Methane and CO_2 were analyzed with a flame ionization detector (FID) and a 90 × 0.3 cm Hayesep D column (Alltech, Deerfield, IL). Nitrogen was used as a carrier gas at the flow rate of 20 mL min^{-1}, and H_2 (25 mL min^{-1}) and hydrocarbon-free air (300 mL min^{-1}) were used as flame gases for CH_4 analysis. Carbon dioxide was measured using the same flame ion detector by first directing gas samples through a methanizer (SRI, Torrance, CA). Oven, FID, and ECD detector temperatures were 50, 150, and 330°C, respectively.

Gas fluxes were calculated from the time vs. concentration data using either linear regression or, when the time vs. concentration data were curvilinear, the algorithm of Hutchinson and Mosier (1981). These data were used to calculate cumulative emissions over the experimental period by linear interpolation and numerical integration using the trapezoid rule. To examine the relationship between %WFPS and N_2O emissions, we estimated the %WFPS values by linear interpolation at times when the fluxes were measured.

Statistical Analyses

Soil effects, N effects, and soil × N treatment interactions on the cumulative GHG emissions were analyzed by two-way ANOVA. Individual contrasts were determined by Fisher's LSD method using SigmaStat software (SigmaStat Version 2.03; SPSS, 1997).

Results

Temporal Dynamics of GHG Emissions

Daily N_2O emission varied from <0.5 to 63 mg N_2O-N m^{-2} d^{-1} (Fig. 1a). Highest emissions were observed from the manure-treated soils. Fluxes from this treatment were highest at the time of manure application. Emissions then rapidly declined over the next 2 to 3 d, but exhibited increases at day 7, following the first watering event. Following subsequent weekly watering events, peaks of N_2O emissions were observed; however, the amplitudes of these peaks decreased over the course of the incubation. Patterns of N_2O emissions in the UAN treatment differed as a function of soil type. In the sandy loam-UAN treatment the N_2O emission pattern was similar to that of the manure treatment; showing an initial peak followed by a general decline, punctuated by additional smaller peaks following watering events. The UAN treatment in the clay soil exhibited N_2O emissions that remained below 10 mg N_2O-N m^{-2} d^{-1} throughout the entire experiment.

Patterns of CO_2 emissions were similar between the two soils (Fig. 1b). Highest CO_2 emissions were observed in the manure treatment, which exhibited peak flux 4 d after manure application. However, decline in CO_2 emission was more rapid in the clay soil than in the sandy loam soil. In both soils, the CO_2 emissions in the UAN treatment were not significantly different than the control treatment.

Methane fluxes were variable, and in every treatment, except for the sandy soil treated with manure slurry, daily emissions were not significantly different from the fluxes in the control plots (Fig. 1c). Methane flux from the sandy loam soil treated with manure slurry exhibited a peak immediately after manure application, and another peak at the first watering event at 7 d. Emissions then decreased over the course of the experiment.

During the course of the experiment water was added at weekly intervals. This resulted in weekly fluctuations in %WFPS that ranged from 40 to 56% in the sandy soil and from 37 to 50% in the clay soil (Fig. 1d). The differences in maximum and minimum %WFPS are due to bulk density differences between the two soils and the differences in soil water content at field capacity. It is interesting to note that trace gas fluxes seem to respond to the fluctuations in %WFPS early in the experiment, but at the later stages of the experiment the amplitudes of the trace gas responses to added water (and increases in %WFPS) diminished. In both soils there is substantial scatter in the relationship between %WFPS and N_2O flux (Fig. 2). Correlation analyses for the individual N treatments

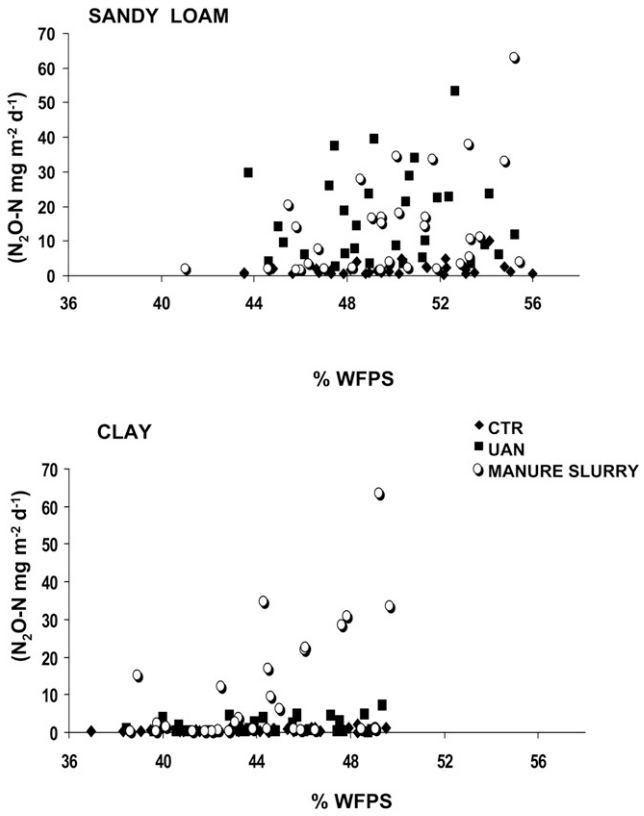

Fig. 2. Relationships between water-filled pore space and N_2O emissions.

within each soil type were not significant and correlation coefficients ranged between 0.010 and 0.209.

Cumulative Greenhouse Gas Emissions

There were significant soil and treatment differences in cumulative N_2O emissions (Table 2). Nitrous oxide emissions in the control treatment of sandy loam and clay soils were not significantly different; however, in the UAN and manure treatments, the sandy loam had significantly greater cumulative N_2O emissions than the clay soil. Within each soil type, there were also treatment differences. Cumulative N_2O emissions in the sandy soil were greatest in the UAN treatment and smallest in the control. In the clay soil higher cumulative N_2O emissions were observed in the manure treatment than in either the UAN or the control treatments. In addition to the soil and treatment differences, the soil × treatment interaction was also significant ($P = 0.029$). After accounting for the cumulative N_2O-N produced in the control treatments, it is calculated that, in the sandy loam, N_2O-N emissions accounted for the equivalent of 3.3% of the UAN nitrogen added and the equivalent of 2.7% of the manure N added with no significant differences between these values (Fig. 3). The clay-UAN treatment lost the least equivalent of 0.24% of the added N, and the clay-manure treatment lost a larger equivalent of 1.84% of added manure N as N_2O (Fig. 3). However, it was less than in the sandy loam-manure and sandy loam-UAN treatments.

Carbon dioxide emissions showed significant soil and treatment effects; however, the soil × treatment interaction was not significant (Table 3). In each treatment, cumulative CO_2 emissions

Table 2. Effects of soil type and N fertility treatment on cumulative N_2O emissions. Values are means of four replicates. Associated standard deviations are shown in parentheses. Probabilities in the right hand column indicate significance of soil effects within the given N treatment. The ANOVA table is presented below, and shows significance of soil x N treatment interaction.

N Treatment	Sandy loam	Clay	*P*
	mg N_2O-N m^{-2}		
Control	85.1 (12.3)	29.6 (12.4)	0.292
UAN	746 (165)	76.6 (9.2)	< 0.001
Manure	628 (59.2)	397 (16.2)	< 0.001
LSD ($P = 0.05$)	***107.4***		

ANOVA

Source of variation	DF	SS	MS	F	*P*
Soil	1	608,825	608,825	116.43	< 0.001
N Treatment	2	914,444	457,222	87.44	< 0.001
Soil x N treatment	2	400,142	200,071	38.3	0.029
Residual	18	94,117	5228		
Total	23	2,017,529	87,719		

were significantly greater in the sandy soil than the clay soil. With respect to N treatment differences, within each soil, cumulative CO_2 emissions from the UAN treatment and the control were not significantly different in either soil. However, in both soils the manure treatment had higher cumulative CO_2 emissions than the other treatments. These elevated CO_2 emissions in the manure treatment were likely due to the added organic C. The dry matter content of the manure used in this experiment was 24 g kg^{-1}. If it is assumed that the C content of the dry matter was 400 g kg^{-1}, the application rate of particulate C in the manure treatments of this study was approximately 106 g C m^{-2}. In the sandy soil the difference in cumulative CO_2–C emissions between the control and manure treatments was 48.5 g C m^{-2}, and in the clay soil the manure treatment produced 38.1 g CO_2–C m^{-2} more than the control treatment. Thus, in the 8 wk of this study the equivalent of 45.5% of the added particulate manure C was lost from the sandy soil, and 35.7% was lost from the clay soil.

Cumulative CH_4 fluxes are presented in Table 4. Only the sandy loam-manure treatment had cumulative CH_4 fluxes that were significantly different from other treatments. The results from ANOVA analysis indicated both a significant soil effect and N treatment effect, as well as a significant soil × N-treatment interaction.

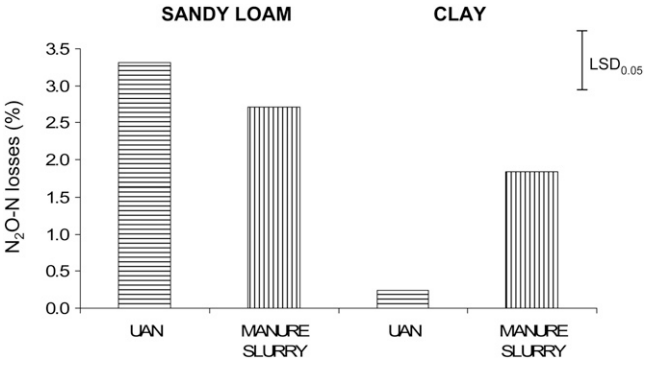

Fig. 3. Percentages of added N lost as N_2O emission from sandy loam and clayey soils fertilized with UAN or manure slurry applied at the rate of 200 kg N ha^{-1}. Error bar indicates $LSD_{0.05}$.

Table 3. Effects of soil type and N fertility treatment on cumulative CO_2 emissions. Values are means of four replicates. Associated standard deviations are shown in parentheses. Probabilities in the right hand column indicate significance of soil effects within the given N treatment. The ANOVA table is presented below showing sources of variation.

N Treatment	Sandy loam	Clay	*P*
	——mg CO_2–C m^{-2}——		
Control	65.5 (7.5)	50.1(5.9)	0.006
UAN	64.4 (4.6)	52.5 (1.9)	0.027
Manure	114 (10.5)	88.2 (5.6)	< 0.001
LSD (*P* = 0.05)	***7.36***		

ANOVA

Source of variation	DF	SS	MS	F	*P*
Soil	1	1908	1908	38.86	< 0.001
N Treatment	2	9944	4972	101.3	< 0.001
Soil x N treatment	2	220	110	2.24	0.135
Residual	18	884	49		
Total	23	12,956	563		

Discussion

Past observations of soil texture effects on N_2O emissions have yielded mixed conclusions. Mkhabela et al. (2006) observed 2.5 times higher N_2O emission from sandy loam soil than from silty clay loam soil. These authors attributed the differences to the fact that the sandy loam had higher pH (5.7) than their silty clay loam soil (pH = 4.7). Conversely, in a summary of studies reporting N_2O emissions in Eastern Canada, Gregorich et al. (2005) describe higher N_2O emissions from a clay soil than a loamy sand. Similarily, Bouwman et al. (2002) reported that fine-textured soils high in SOC showed higher N_2O emissions than less fertile soils. However, Bouwman (1996), in an analysis of N_2O emissions from fertilized and unfertilized lands, reported on conflicting observations of the effects of soil texture on N_2O emissions. He attributed the observations of higher N_2O emissions in heavy textured soils to enhanced anaerobic conditions, whereas in lighter textured soils, it was speculated that weather conditions dominated any texture effects. Clearly, soil texture influences several factors that control N_2O emissions, including aeration, organic C availability, and N availability. Thus, depending on the interplay of these controlling factors at the specific sites of N_2O production in soil, it is not unreasonable to expect varied effects of texture on N_2O emissions.

Table 4. Effects of soil type and N fertility treatment on cumulative CH_4 fluxes. Values are means of four replicates. Associated standard deviations are shown in parentheses. Probabilities in the right hand column indicate significance of soil effects within the given N treatment. The ANOVA table is presented below, showing sources of variation.

N Treatment	Sandy loam	Clay	*P*
	——mg CH_4–C m^{-2}——		
Control	–6.4 (19.6)	–5.9 (4.9)	0.98
UAN	13.6 (32.6)	–8.8 (31.1)	0.20
Manure	113 (20.3)	15.2 (21.2)	< 0.001
LSD (*P* = 0.05)	***34.85***		

ANOVA

Source of variation	DF	SS	MS	F	*P*
Soil	1	9624	9624	17.48	< 0.001
N Treatment	2	23,699	11,849	21.53	< 0.001
Soil x N treatment	2	10,685	5342	9.71	0.001
Residual	18	9908	550		
Total	23	53,917	2344		

The sandy loam soil of our study exhibited higher emissions of N_2O than the clay soil. There could be several texture-related factors contributing to this result. In our protocol, the water contents of our experimental units were adjusted to field capacity at weekly intervals. As a result of the bulk density differences between our two soils, there were differences in %WFPS, with the sandy soil having higher average % WFPS than the clay soil. Percent water-filled pore space has been used as an indicator of soil aeration state (Linn and Doran, 1984). Davidson (1991) provided a general relationship between %WFPS and N_2O emissions. This relationship predicts net N_2O emissions between %WFPS values of 30 and 90, with a peak N_2O production occurring at approximately 65%WFPS. Within a given soil type, fertility regime, and cropping system this idealized relationship may be valid, but generalizations across soils and management systems are likely to be poor. Indeed, some recent studies report that N_2O emissions do not exhibit a peak in the range of 60 to 65% WFPS, but rather increase up to 80 to 90% WFPS (Adviento-Borbe et al., 2006; Maljanen et al., 2007). Unlike other laboratory studies investigating %WFPS and N_2O production, in our study %WFPS was not held constant. In the early stages of our incubation, increases in N_2O emissions were observed after water additions (and resulting increases in %WFPS); however, the amplitudes of these N_2O peaks following water addition diminished with time. Thus, over the course of our experiment we did not observe a strong relationship between %WFPS and N_2O emissions in either of our soils. This diminished response of N_2O to changes in %WFPS in the later stages of our incubations indicates that factors other than soil water content were limiting N_2O emissions.

Differences in N availability in our two soils may have been a factor controlling N_2O production. It is known that the composition of the soil mineral fraction as well as soil organic C can impact ammonium absorption (Bremner, 1959; Burge and Broadbent, 1961). In addition, it has been shown that decreased N availability due to ammonium fixation by soil colloids can reduce the activity of nitrifying bacteria (Hommes et al., 1998). De Visscher et al. (1998) demonstrated that increasing inputs of NH_4^+ to soil with low CEC was associated with higher N_2O emissions. They concluded that soils with high CEC facilitated immobilization of NH_4^+ at cation exchange sites, whereas in soils with low CEC, higher concentrations of free NH_4^+ were available. A recent study investigated the adsorption of manure NH_4^+ and ammonium sulfate NH_4^+ in two soils having different CECs (Fernando et al., 2005). These authors found that in a clay soil with a CEC of 20 cmol kg^{-1} and clay content of 20%, sorption of swine manure slurry NH_4^+ was initially more rapid than $(NH_4)_2SO_4$–NH_4^+, but after 80 h equilibrium sorbed NH_4^+ concentrations of both materials were approximately equal. Measured NH_4^+ adsorption maxima for each applied N material in each soil showed that the clay soil had NH_4^+ adsorption maxima of 1000 and 909 mg NH_4^+ kg^{-1} soil for the manure and $(NH_4)_2SO_4$ treatments, respectively. The NH_4^+ adsorption maxima in the sandy soil (CEC = 11 cmol kg^{-1}, 8% clay) were lower (625 mg NH_4^+ kg^{-1} soil for the manure treatment

and 217 mg NH_4^+ kg^{-1} soil for the $(NH_4)_2SO_4$ treatment). The greatest difference between their soils was in the $(NH_4)_2SO_4$ treatment. This was thought to be due to the fact that dissolved organic C in the manure served to promote NH_4^+ adsorption. Our results can be explained in the context of these authors' observations. Ammonium N sorption predicted from the texture-organic matter interactions described by Nishantha et al. (2005) would have resulted in N availability in our treatments as follows: sandy loam-UAN > sandy loam-manure > clay-UAN > clay-manure. This order of predicted N availability was reflected in our observations of cumulative N_2O loss in our sandy loam soil, but not in our clay soil. We observed cumulative N_2O losses in the order: sandy loam-UAN > sandy loam-manure > clay-manure > clay-UAN (Table 2). In our UAN treatment cumulative N_2O losses were approximately ninefold greater in the sandy loam than in the clay soil. This effect could be due, at least in part, to the higher NH_4^+-N adsorption in the clay, and thus, lower N availability. However, based on the results of Nishantha et al. (2005) it should be predicted that the lowest NH_4^+ availability would have been present in our clay-manure treatment. Yet this treatment had cumulative N_2O emissions that were greater than the clay-UAN treatment. The unknown factor possibly contributing to this apparent discrepancy is the susceptibility of the sorbed organo-NH_4^+ complexes to microbial decomposition. Also, it should be recognized that NH_4^+ comprised only 44% of the added N in the manure slurry treatment, thus the organic N in the manure may have been a source for N_2O production. We observed enhanced CO_2 production in the manure slurry treatments relative to the UAN treatments (Table 4), indicating that organic material in the manure was being decomposed. It is likely that some of this N released might have been available to nitrifying or denitrifying organisms.

Differences in N availability within our soil-treatment combinations are also consistent with our observed differences in CH_4 emissions. Net CH_4 emissions from soil are the result of two competing processes; methanogenesis and methane oxidation. Several studies have shown that CH_4 oxidation is lower in agricultural soils than in natural systems (Bender and Conrad, 1993; Dobbie and Smith, 1996; Powlson et al., 1997). This effect, in part, is thought to be due to fertilizer N inhibition of CH_4 consumption activity in arable soils (Steudler et al., 1989; Mosier et al., 1991; Bronson and Mosier, 1994). Indeed, NH_4^+ has been reported to be a competitive inhibitor of CH_4 oxidation (Whittenbury et al., 1970; Hyman and Wood, 1983; Jones and Morita, 1983). However, the response of methane oxidation to N additions may be moderated by soil texture. It has been observed that inhibition of methane oxidation in response to N additions was greater in a sandy loess soil than in a clay soil (Hütsch et al., 1993; Hütsch, 1996). Similarly, the differences in CEC in our sandy and clay soils could have impacted free NH_4^+ concentrations that, in turn, could have had a differential effect on CH_4 oxidation (De Visscher et al., 1998). However, net CH_4 emissions are also dependant on CH_4 production. Manure applications to soil, in addition to providing an inoculum of methanogenic bacteria, stimulate O_2 consumption and facilitate the development of anaerobic conditions. Manure slurry has been observed to promote CH_4 fluxes immediately after application to the soil (Sommer et al., 1996; Chadwick et al., 2000; Chan and Parkin, 2001a; Sherlock et al., 2002) and in response to rainfall events following manure applications (Chan and Parkin, 2001a). Thus, in combination with the inhibitory effects of manure NH_4^+ on methane oxidation, stimulation of CH_4 production in soil by manure would tend to increase net CH_4 emissions. In our study, we observed significant CH_4 fluxes from the sandy loam soil in response to manure application. In our other treatments cumulative CH_4 fluxes were not significant from zero.

Conclusions

Manure additions to the sandy loam soil significantly increased CH_4 emissions. However, methane emissions from other treatments were not significantly different from zero. We observed a significant soil effect on cumulative N_2O emissions. The lack of a significant relationship between %WFPS indicates that N availability may be a primary mechanism controlling N_2O emissions. We also observed a significant soil type × N treatment interaction. We speculate that differences in NH_4^+ fixation between the two soils could be a factor controlling N availability for N_2O production. The significant soil × N treatment interaction may have relevance to current efforts aimed at prediction of N_2O emissions.

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