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AP42 Section:	9.2.1
Reference:	21
Title:	F.C. Thornton and R.J. Valente, "Soil Emissions Of Nitric Oxide And Nitrous Oxide From No-Till Corn, Soil Science Society of America Journal, 60:1127-1133, 1996.

WATSON 5/21/97 AP-42 #21

Soil Emissions of Nitric Oxide and Nitrous Oxide from No-till Corn

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ABSTRACT

Fertilized agricultural soils can be a significant source of emissions of NO and N_2O into the atmosphere. This study was conducted to determine the influence of N rate on the emissions of these gases in a no-till corn (*Zea mays* L.) crop grown in western Tennessee. The influence of N rate was assessed for a 210-d period on replicated plots receiving 0, 140, and 252 kg N ha^{-1} (0N, 140N, and 252N) as ammonium nitrate (AN). Plots were located on a Routon silt loam (fine-silty, mixed, thermic Typic Ochraqualf) at the West Tennessee Agricultural Experiment Station in Jackson, TN. Gas fluxes were measured by static chamber boxes located on plots. The measurement technique was automated and replicate chamber estimates were made eight times daily for the entire study period. Fertilizer application significantly affected both NO and N_2O emission rates. The cumulative N_2O-N lost from the fertilizer treatments was from 10 to 20-fold that of NO. On an areal basis, the 140N treatment emitted 4.23 kg N_2O-N and 0.19 kg ha^{-1} of NO-N whereas the 252N treatment emitted 6.56 kg N_2O-N and 0.50 kg ha^{-1} NO-N. Soil parameters of water-filled pore space (WFPS), NO_3^- and NH_4^+, were correlated with N_2O emissions but only soil NO_3^- was correlated with NO flux. Our data, and more recent data in the literature, suggest that N_2O emissions from fertilized soil may be considerably higher than previously thought. Emissions of N_2O were 2.6 to 3.0% of the fertilizer amounts applied. These higher emissions may, in part, explain some of the reason for the shortfall in the global N_2O budget.

NITROGEN OXIDES (N_2O, NO, and NO_2) are important trace gases in terms of the atmospheric chemistry of both the stratosphere and troposphere. As a greenhouse gas, N_2O tends to warm the lower atmosphere and the earth's surface by its absorption and re-emission of radiation in the atmosphere (Bouwman, 1990). Nitric oxide ($NO_x = NO + NO_2$) is important in controlling troposphere ozone levels through its reaction with volatile organic compounds in the atmosphere. Tropospheric ozone is a pollutant that negatively impacts forest and crop growth, as well as human health. Despite efforts to control NO_x emissions, tropospheric ozone concentrations have continued to increase during the past decade (National Research Council, 1992).

Prior to industrialization, the main source of these emissions into the atmosphere was biological activity related to the soil processes of nitrification and denitrification. A number of key factors such as soil moisture, C content, and soil aeration affect emission processes (Aulakh, 1992). The magnitude of these emissions however, is also dependent on soil levels of NH_4^+ and NO_3^-, and with the advent of modern agriculture and the increased use of mineral fertilizers there has been an increase in soil emissions of these gases. Agriculture is presently estimated to contribute from 65 to 80% of the total anthropogenic N_2O, or up to 2 Tg N_2O-N yearly (International Panel on Climate Change, 1990; Iserman, 1994). Cropland is also thought to be a major emitter of NO. Davidson's (1991) analysis, albeit made from a limited data base, suggests that 30% of the 20 Tg of the total NO budget is attributable to soil emissions from agricultural activity.

Although only a few studies have been performed to determine the influence of N fertilizer application on the simultaneous emissions of N_2O and NO, they consistently demonstrate that fertilization increases emissions of nitrogen oxides (see Williams et al., 1992). The estimates of global emissions span a wide range and indicate that the NO/N_2O ratios of emissions vary widely among studies. For example, Galbally et al. (1987) reported that <0.002% of applied urea was lost as NO, whereas Hutchinson and Brams (1992) reported that 3.2% of AN applied to a bermudagrass (*Cynodon dactylon* [L.] Pers.) pasture was lost as NO after 9 wk. Typically, soil efflux of NO and N_2O does not result in substantial N losses compared with the amount of fertilizer applied, but these emissions are important in atmospheric chemical reactions in both the troposphere and stratosphere (see Williams et al., 1992). Shepherd et al. (1991) observed very large losses where 11% of fertilizer N was lost as NO and 5% was lost as N_2O over 5 mo. The wide divergence in the estimates of NO and N_2O emitted from soils is a result of both the large temporal and spatial variability associated with the ecological processes that control the emissions of these gases as well as differences associated with the measurements and the measurement methodologies used. Environmental factors such as precipitation events can also significantly change efflux patterns by changing soil redox potential or affecting soil gas diffusivity and microbial activity (e.g., nitrification and denitrification) and subsequent N gas production and efflux (Valente and Thornton, 1993). Coefficients of variation associated with emissions estimates for these gases are typically between 100 and 300% (Williams et al., 1992; Valente and Thornton, 1993; Valente et al., 1995). To some extent, this variation is due to the fact that discrete point estimates of gaseous emissions have been used to calculate flux. It is only recently that quasi-continuous measurements for extended periods of time have been made to estimate fluxes of NO and N_2O (Loftfield et al., 1992; Valente and Thornton, 1993). Many of the point estimates of soil trace gas efflux are not adequate to assess seasonal or annual losses of NO and N_2O because continuous measurements are needed (Aulakh, 1992).

To address the need for daily flux measurements to estimate NO and N_2O emissions from fertilized soils, we continuously measured NO and N_2O emissions over a 210-d period from no-till corn fertilized at three N rates. In addition to gaseous emissions, we monitored

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Published in Soil Sci. Soc. Am. J. 60:1127–1133 (1996).

Abbreviations: AN, ammonium nitrate; WFPS, water-filled pore space; ICP, inductively coupled plasma emissions spectroscopy.

soil chemical and physical properties in an attempt to understand mechanisms that controlled these effluxes.

MATERIAL AND METHODS

Site

Emissions estimates were made at the West Tennessee Agricultural Experiment Station in Jackson, TN (35°37′N, 88°50′W) from 27 Apr. through 30 Nov. 1993. The soil is a Routon silt loam derived from loess deposits 2 to 5 m thick, overlaying Coastal Plain sediments. Table 1 lists selected properties of the soil. The field used for this experiment has been in no-till corn for the past 4 yr. The long-term climatic data (30-yr average) for the site indicate that mean annual rainfall is 1320 mm, with a mean annual temperature of 17°C, and approximately 193 frost-free days during the growing season (Springer and Elder, 1980).

A randomized complete block design with three N rates and three replications was used to assess N gas efflux. The three fertility treatmentswere 0N, 140N, and 252N applied as AN on 26 Apr. 1993. Corn (Pioneer 3925) was planted 21 Apr. 1993 in 76 cm rows at 39 000 seeds ha^{-1}. The N rates were those recommended dryland corn for this soil. 140 kg N ha^{-1}, and the 252 kg N ha^{-1} rate was the upper-end N rate for corn grown on this soil under irrigation (irrigation was not used in our experiment). Four measurement plots were randomly established within each treatment × block combination for a total of 36 measurement locations. At the start of the experiment on 27 Apr. 1993, two of the four sampling locations within each treatment in Block I were used to make measurements of NO and N_2O. The following week the two locations that were not chosen in Week 1 were used to make measurements. The following week the same procedure was used for the second experimental block to locate sample locations. This procedure was repeated, starting in Week 5 for the third replicate block, after which the same selection sequence was repeated among blocks until the end of the experiment (30 Nov. 1993). Thus, each chamber replicate within a measurement plot for a treatment was sampled every 3 h for approximately 1 wk (n = 56) and each block was sampled five time (two consecutive weeks at a time) throughout the experiment.

To minimize soil disturbance associated with the relocation of the measurement chambers throughout the experiment, the six frames used to make measurements (see below) were placed in the ground 1 wk before the measurements were made. This enabled quick placement of chambers onto frames with minimal soil disturbance.

Yield estimates were obtained by harvesting two 4.65-m sections of row in the middle of each plot. Plots were harvested by hand and grain yields are expressed on a 10% moisture basis.

Table 1. Soil characteristics of the 0- to 15-cm layer of a Routon silt loam prior to the start of the experiment.

Texture, %	
Sand	24
Silt	52
Clay	24
Bulk density, Mg m^{-3}	1.61 (0.14)†
pH	5.75 (0.23)
NH_4^+, mg kg^{-1}	4.3 (0.4)
NO_3^-, mg kg^{-1}	16.7 (2.4)
P, mg kg^{-1}	113 (17)
Ca, mg kg^{-1}	1061 (211)
Mg, mg kg^{-1}	42 (7)
K, mg kg^{-1}	114 (26)

† Mean with standard deviation in parentheses.

Gas Measurements

Measurements of NO and N_2O emissions were made with a static-chamber technique as previously described by Valente and Thornton (1993). Briefly, aluminum chambers (L by W by H = 76.2 by 45.7 by 20.3 cm) opened and closed pneumatically over an aluminum frame driven into the ground to a depth of 20 cm. Multiport sequencing of chamber air samples allowed withdrawal of air samples from different chambers at 3-min intervals. In contrast to the method described by Valente and Thornton (1993) the Teco Model 46 N_2O gas correlation instrument (Thermo Environmental Instruments, Franklin, MA) was placed upstream of the Teco Model 42 used to determine NO. The pump of the Teco Model 46 was bypassed and the Teco Model 42 pump was used to withdraw sample air from chambers. This was done to reduce the amount of sample withdrawn from chambers since the Model 42 withdrew only 0.7 L min^{-1} rather than the 1 L min^{-1} for the Teco Model 46. At 3-min intervals, NO and N_2O measurements were performed and the emission rate and deposition rate calculated from the change in gas concentrations during the measurement period. The calculation of NO efflux rate was identical to that described by Valente and Thornton (1993) in which a first-order differential equation of the form

$$\frac{d[NO]}{dt} = k_1 - k_2[NO]$$

was used to calculate a gross emissions rate; where k_1 is a constant emission rate term, k_2 is a loss term due to deposition within the chamber, and [NO] is NO concentration. Emission estimates for N_2O were calculated using a linear model. The lower limit of sensitivity for NO–N emissions was 0.002 ng N m^2 s^{-1} and the lower limit for N_2O was 0.02 ng N m^2 s^{-1}. At the end of the measurement period, all six chambers were opened and remained open for 2 h until the next measurement cycle began. This sequence, of automated chamber closure and the attendant sampling of NO and N_2O concentrations with time within chambers, was repeated every 3 h, 24 h a day, throughout the experiment. A Campbell Scientific Model CR7 data logger (Campbell Scientific, Logan, UT) was used to control the opening and closing of chambers and collect and store data gathered during each measurement cycle.

Weekly calibration checks were made for both N_2O and NO analyzers. Certified gases (Scott Specialty Gases, Plumsteadville, PA) of NO (19.24 μL L^{-1}) and N_2O (80.6 μL L^{-1}) were used to make multipoint calibrations. Coefficients of determination for standard curves for both gases was consistently 0.995 or greater throughout the study. Several times during the study, line loss tests were performed by introducing standard gases directly into the sample lines of chambers. Gas loss throughout the 40-m sample lines averaged approximately 5% and was considered negligible for the calculations of emissions rates. These finding are consistent with a previous test of the system (Thornton and Valente, 1992, unpublished data).

Ancillary Measurements

Soil temperature at 5- and 15-cm depths was continuously monitored by means of a thermocouple during the study. Gravimetric water content determinations, based on oven-dry soil (105°C), for the 0- to 15-cm depth were made weekly. A soil moisture retention curve was determined using triplicate undisturbed soil cores for 0.03, 0.06 and 0.1 MPa tensions; tensions of 0.3, 0.5, and 1.5 MPa were determined on disturbed

samples. The weekly determinations of gravimetric soil moisture were used to determine WFPS from the relationship where:

$$\text{WFPS} = \frac{\text{Grav.H}_2\text{O} \times \text{Bulk Density}}{\text{Total Porosity}}$$

$$\text{where porosity} = 1 - \left(\frac{\text{Bulk Density}}{2.65}\right)$$

Rainfall was recorded using a tipping bucket rain gauge (Climatronics, Bohemia, NY). Soil samples were also collected weekly from the top 15-cm layer for chemical analysis of NO_3^- and NH_4^+. Soils were extracted with 2 *M* KCl and NO_3^- and NH_4^+ determined by titration. Cation determinations were made using Mehlich 3 procedures (Mehlich, 1984) followed by ICP determinations.

Statistical Analysis

Gas flux data were analyzed using the general linear models procedure in the Statistical Analysis Systems for personal computers (SAS Institute, 1993). Differences among treatment means were evaluated using a Bonferroni *t*-test. Mean emission rates for NO and N_2O were calculated for each treatment from the entire 210-d experimental period. Correlation coefficients between soil parameters and trace gas efflux were estimated for the combined N treatments and the daily average efflux values for NO and N_2O for the days on which the soil parameters were measured. A 0.05 probability level was used to determine significant differences.

RESULTS AND DISCUSSION

Fertilizer application significantly affected emissions of both N_2O and NO (Table 2, Fig. 1–3). Emissions of N_2O were more episodic in nature than NO, which was emitted at a somewhat uniform rate during the course of the study. The loss of fertilizer N during the study period was significantly influenced by N application rate for both N_2O ($F = 73.5$, $P = 0.0001$) and NO ($F = 184.1$, $P = 0.0001$) The efflux of N_2O was 4.23 kg of N from the 140N treatment and 6.56 kg N in the 252N treatment (Table 2). Average emissions rates with no N application (control plots) were 10.4 and 3.5 ng N m^2 s^{-1} for N_2O and NO, respectively, resulting in a loss of 1.98 kg N_2O–N and 0.63 kg NO–N during the 210-d study. Emission rates of NO from the fertilized treatments were considerably lower than we measured in past studies on fertilized row crops. On a similar silt loam soil in middle Tennessee that had received 100 kg N ha^{-1}, we previously reported a midsummer rate of 24 ng NO–N m^2 s^{-1} (Valente and Thornton, 1993). In a separate study on a red clay soil in Alabama, a cotton (*Gossypium hirsutum* L.) crop receiving 112 kg N ha^{-1} had an average emission rate of 17 ng NO–N m^2 s^{-1} (Valente et al., 1995) or an emission rate that was approximately five times higher than the present study. Williams et al. (1988) reported an even higher mean NO emission rate of 94 ng N m^2 s^{-1} for corn in Pennsylvania. Our results are more similar to those reported by Anderson and Levine (1987), who reported an average soil NO efflux rate of approximately 6.5 ng NO–N m^2 s^{-1} for plots receiving 163 kg N ha^{-1}. However, in as much as variation in emission rates across a single site is typically threefold (Davidson, 1991), it is not unreasonable to expect that variation among sites would have a variation that is 10-fold due to differences in soil texture and attendant water-holding capacity (i.e., WFPS), timing and uptake of soil N, and cultural practices such as fertilizer type and placement that may affect emissions.

Except for a few instances of significant denitrification

Table 2. Mean soil efflux rates and total losses of N as N_2O and NO related to N fertilization rate for a 210-d period.

Treatment	Emission rate	N loss	N loss
kg N ha^{-1}	ng N m^{-2} s^{-1}	kg N ha^{-1}	% of applied N
		N_2O	
0	10.4 (34.9)†	1.98 (–)‡	–
140	28.1 (91.8)	6.21 (4.23)	3.02
252	38.4 (112.7)	8.54 (6.56)	2.60
		NO	
0	3.5 (4.6)	0.63 (–)	–
140	4.5 (5.3)	0.81 (0.19)	0.13
252	6.4 (6.5)	1.13 (0.50)	0.20

† Mean rate and standard deviation, $n = 2705$.
‡ Total N loss was calculated. N loss attributable to fertilizer application indicated in parentheses.

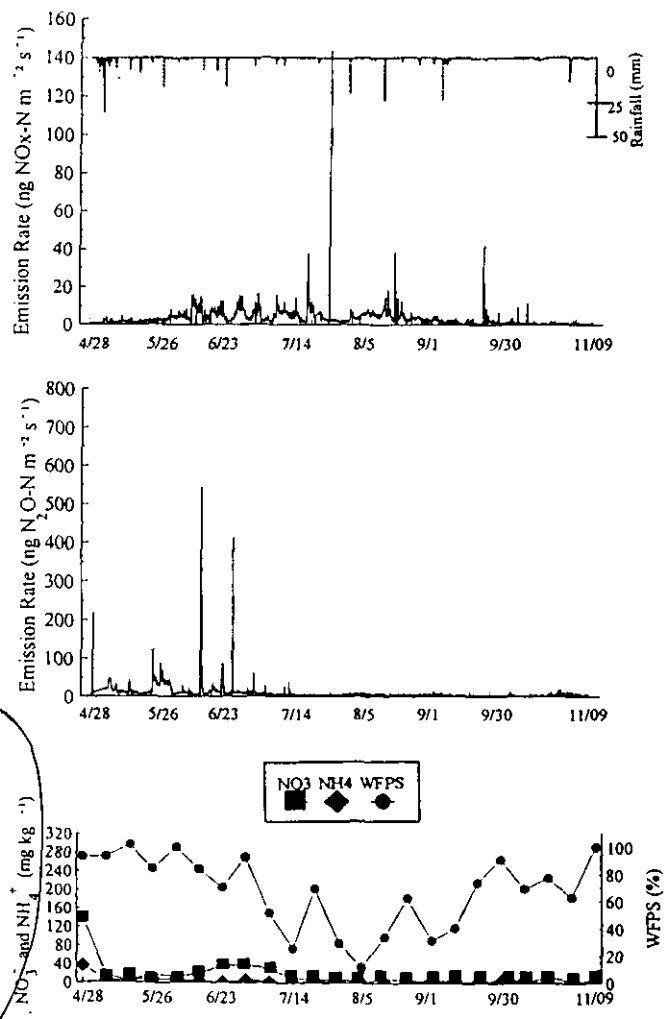

Fig. 1. Data for a 210-d period for no-till corn receiving 0 N kg ha^{-1}: (a) NO emission rate and rainfall, (b) N_2O emission rate, and (c) soil NO_3^-, NH_4^+, and water-filled pore space (WFPS) for the 0- to 15-cm depth.

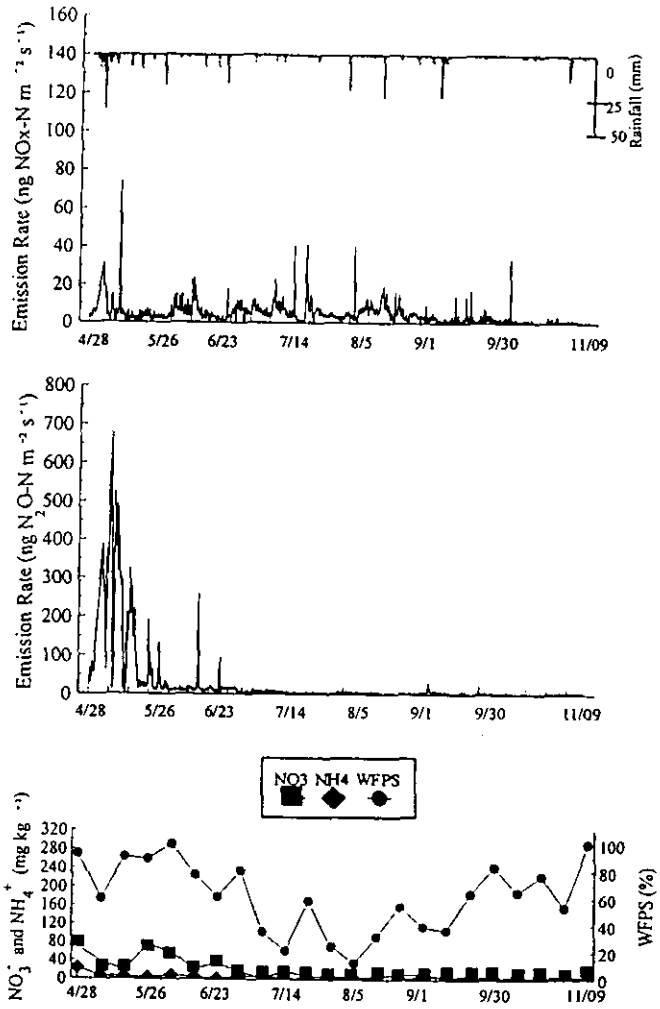

Fig. 2. Data for a 210-d period for no-till corn receiving 140 N kg ha^{-1}: (a) NO emission rate and rainfall, (b) N_2O emission rate, and (c) soil NO_3^-, NH_4^+, and water-filled pore space (WFPS) for the 0- to 15-cm depth.

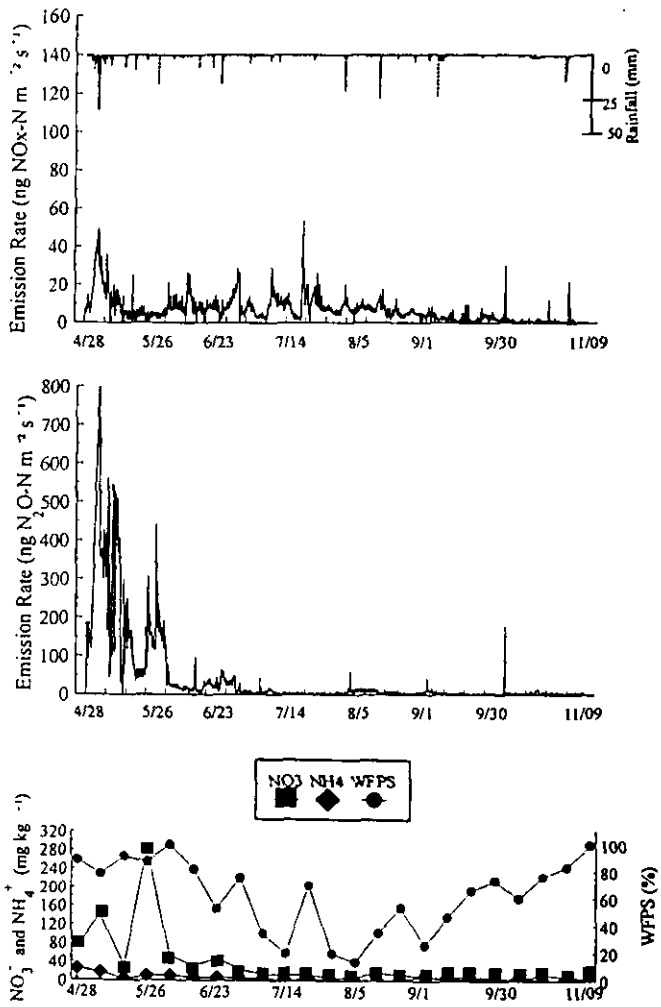

Fig. 3. Data for a 210-d period for no-till corn receiving 252 N kg ha^{-1}: (a) NO emission rate and rainfall, (b) N_2O emission rate, and (c) soil NO_3^-, NH_4^+, and water-filled pore space (WFPS) for the 0–15 cm depth.

activity later in the season, most of the N_2O emission occurred during the first month of the study. The typical, usually immediate, increase in N_2O emissions following N application (Breitenbeck et al., 1980; Duxbury et al., 1982) was evident in this study (Fig. 1–3). The N loss of N_2O from nonfertilized soil was considerably higher than that cited by Eichner (1990), who reported an average emission of 0.8 kg N ha^{-1} yr^{-1} for nonfertilized soils. The amount of fertilizer-derived N_2O emissions in the present study is approximately sevenfold the average loss of 0.44 kg N ha^{-1} reported for AN in the review by Eichner (1990). The substantially higher values in our study, compared with the average given by Eichner (1990), may be related to a number of factors including soil type, rainfall, timing of fertilizer application, and management practices, which can strongly influence emissions on a site-specific basis. These differences may also result from using average daily emission rates that are calculated from discrete point samples that are oftentimes only made once a week. Annual estimates require methods that use both long-term and highly time-resolved determinations of flux (Loftfield et al., 1992) and this study attempts to address this need by having replicated measurements every several hours over 7 mo.

Soil emissions of NO and N_2O were not affected by harvesting. We found no increase in emissions after the 9 September harvest. We had speculated that increased C supply due to harvest residue would promote denitrification; however, soil moisture was very low and apparently little denitrification occurred. Hutchinson and Brams (1992) reported a stimulation in soil NO emission, but not N_2O, after harvesting bermudagrass and speculated that increased soil temperature increased soil efflux of NO.

More recent studies of N_2O and NO emissions from soils, not available at the time of Eichner's (1990) review, show higher emissions than she reported. Hutchinson and Brams (1992) reported that NO emissions during a 9-wk study resulted in a 2.37 kg N ha^{-1} loss while N_2O accounted for 0.35 kg N ha^{-1}. Recent studies by Bronson et al. (1992) on conventially tilled corn over a 2-yr period measured N_2O losses of 3.2 and 1.6 kg N, similar to this study. Studies in Canada (Shepherd et al., 1991) and in Sweden (Hansen et al., 1993) measured losses

Table 3. Correlation coefficients for NO and N_2O soil emissions rates with soil N and moisture at Jackson, TN.

	NO emission rate	N_2O emission rate
Water-filled pore space	0.14 (NS)†	0.32 (0.0001)
NO_3^-	0.25 (0.03)	0.39 (0.0009)
NH_4^+	0.11 (NS)	0.30 (0.10)

† Parenthetical values are the probability levels associated with the correlation; NS indicates statistical nonsignificance.

up to 5.3% of the applied AN as N_2O. In the Canadian studies, 11% of the fertilizer N applied was lost as NO and efflux rates as high as 74 ng NO–N m^2 s^{-1} were reported (Shepherd et al., 1991). Although we measured NO efflux rates as high as 143 ng NO–N m^2 s^{-1} in our study, >95 % of all NO measurements, or 7584 separate measurements, indicated the NO emission rate was below 14 ng N m^2 s^{-1} Recent studies indicate greater loss rates of N via N_2O efflux from soil, but our study, to the best of our knowledge, has some of the highest N_2O emissions rates reported. Values calculated from the work of Eichner indicate an average emission rate of 34 ng N_2O–N m^2 s^{-1}, but Clayton et al. (1994) reported values as high as 1766 ng N m^2 s^{-1}. Emission values for N_2O as high as 994 ng N m^2 s^{-1} were measured in our study for the 252N treatment.

Table 3 shows the correlation coefficients of NO and N_2O efflux with soil WFPS, soil NO_3^- and NH_4^+. These parameters have been reported to strongly influence trace gas emissions from soil (Davidson, 1991). During the first 55 d of the study (through 30 June), WFPS averaged 85% among the three treatments, the same time frame in which most of the N_2O was emitted from the soil. The high values for WFPS are reflected in the above-average rainfall for this period; precipitation was 20% above normal, or 50 mm in excess of the 30-yr average (National Oceanic and Atmospheric Administration, 1993). Denitrification becomes increasingly important as WFPS exceed 60% (Davidson, 1991) and our data support this observation. With decreasing rainfall in July, WFPS was typically <60% (the lowest value recorded was 14% on 5 August; see Fig. 1–3) and denitrification became less important as a factor in N loss. Precipitation for June through August was 50% below normal, or 192 mm below the 30-yr average (National Oceanic and Atmospheric Administration, 1993). This moisture deficit severely limited grain filling and yields for all treatments; yields for the 0N, 140N, and 252N treatments were 2.62, 6.22, and 6.56 Mg ha^{-1}, respectively.

Soil NO_3^- and NH_4^+ concentrations were correlated with N_2O emissions but only soil NO_3^- was significantly linked to NO emissions (Table 3). The literature reports of correlations between soil parameters such as NO_3^-, NH_4^+, and WFPS and emissions of NO and N_2O are conflicting (Williams et al., 1992). Our results are similar to those of other researchers (Aulakh et al., 1992; Bronson and Mosier, 1993) who have reported weak or no relationship between gaseous efflux and soil chemical and physical properties. Slemr and Seiler (1991) contend that although ample data exist showing that NO and N_2O emission rates are dependent on soil chemical and physical properties, it is usually just one variable that

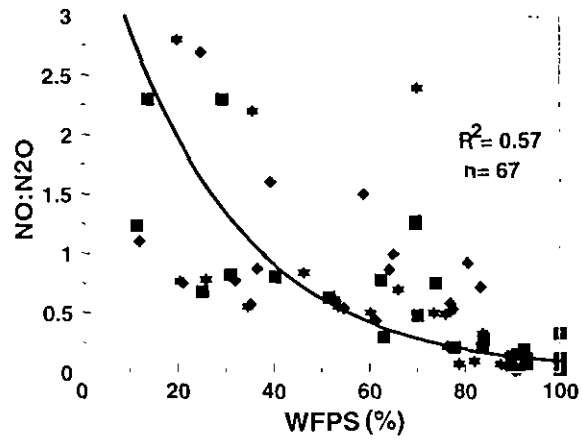

Fig. 4. Relationship between NO/N_2O emission rate ratio and soil water-filled pore space (WFPS) during the study period.

dominantly influences the emission rate. These researchers further maintained that this masking of variables at a particular point in time, location, or study is, in part, responsible for the oftentimes conflicting reports in the literature concerning the importance of a particular variable in influencing trace gas emissions. Several researchers have speculated that C availability, and factors controlling C supply to microbial populations, may be more important in regulating denitrification than has customarily been thought, and most studies dealing with trace gas emissions have not dealt with this important aspect of trace gas efflux (Groffman and Tiedje, 1991; Wheatley and Williams, 1989).

Although individual soil properties were poorly correlated with N gas emissions, WFPS did a reasonable job (R^2 = 0.57) of predicting the NO/N_2O ratio (Fig. 4). These data support recent work (Davidson, 1993; Riley and Vitousek, 1994) indicating that this ratio is low when the soil is wet, typically in the spring, as in our study, and N_2O emissions from denitrification dominate. Conversely, ratios increase as the soil becomes drier and nitrification and NO efflux become more important.

SUMMARY

This experiment demonstrated that loss of fertilizer N as N_2O was far greater than loss as NO; N loss via N_2O was 10 to 20-fold more than that of NO for the study. Nitrous oxide fluxes were positively correlated to WFPS, NO_3^-, and NH_4^+ but only soil NO_3^- values were correlated to NO flux. The ratio of NO/N_2O emissions was reasonably well correlated (R^2 = 0.57) with WFPS and supports the conceptual hole-in-the-pipe model of Firestone and Davidson (1989). In their model the sum of NO + N_2O emissions is related to the N applied, whereas the relative proportion of NO vs. N_2O is linked with WFPS. This experiment suggests that the magnitude of fertilizer-derived emissions estimated by Eichner (1990) from AN may be low. This contention is also supported by recent studies (Hansen et al., 1993; Shepherd et al., 1991) that also indicate much higher emissions rates and subsequent N from applied AN. Using an average emission rate of 40 mg N_2O–N from each kilogram of applied N, Eichner

(1990) estimated that 64.4 Gg N_2O-N was associated with the application of 16.2 Tg AN worldwide for the year 1988. Using the mean efflux value of the two fertilizer rates in our experiment to extrapolate, we would estimate that approximately 455 Gg N_2O-N would be released from AN-fertilized soil annually. Admittedly, extrapolation of emissions data from one location for 1 yr on a fine-textured soil like the one used in this study is fraught with problems; however, this higher estimate in N_2O release may be of importance in light of the fact that global N_2O budgets cannot account for some 7 to 8 Tg of emissions. It is also probable that the techniques used in this experiment and those of Loftfield et al. (1992) better characterize temporal emissions patterns, which can affect budget estimates for a given biome. However, there are still a number of key areas yet to be adequately addressed in attempting to resolve emission estimates on a global basis. Several of those requirements are mentioned in the recent modeling effort of Yienger and Levy (1995), who point to the need to (i) better characterize agricultural emissions from Europe and northern China and (ii) better estimate emissions from tropical grasslands.

The present work also demonstrates an improved sampling technique to allow for quasi-continuous sampling of both N_2O and NO gas efflux from ecosystems. We are currently modifying this system to accommodate greater replicate sampling to improve our estimates of soil efflux for future study sites.

ACKNOWLEDGMENTS

The authors gratefully acknowledge the assistance of Charlie McDuffie, Jr., during the study. Ms. Cassie Wylie's work in the statistical analysis of data is gratefully acknowledged, as well as the efforts of Lynn Humes in preparing figures. The comments of Dr. Bernie Byrnes on an earlier draft of this manuscript are gratefully acknowledged. We also would also like to thank the staff at the West Tennessee Agricultural Experiment Station, particularly Dr. Jim Brown, Dr. Don Tyler, and Mrs. Karen Gibson.

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Determination of Microbial Biomass and Nitrogen Mineralization following Rewetting of Dried Soil

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ABSTRACT

Routine soil testing procedures that are rapid and precise are needed to evaluate agricultural surface soils for their potential to mineralize C and N. Our objectives were to determine the optimum preincubation time after rewetting of dried soil for estimating soil microbial biomass (SMB) and to identify a quick, reliable biochemical predictor of soil N mineralization potential. Biochemical determinations of SMB were screened on a Weswood silty clay loam (fine, mixed, thermic Fluventic Ustochrept) having five levels of soil organic C (SOC) as a result of long-term management. Determinations used (i) field-moist soil and (ii) soil that was air dried, rewetted, and preincubated for 0.2, 1, 3, 6, 10, and 15 d. Biochemical determinations included arginine ammonification, substrate-induced respiration (SIR), cumulative C and net N mineralization, and SMBC using the chloroform fumigation-incubation (CFI) method. Preincubation periods of 1 and 15 d prior to fumigation gave estimates of SMBC using CFI most similar to those determined on field-moist soil. Arginine ammonification and SIR determinations on dried soil were highly variable, making longer preincubation periods necessary. Carbon mineralization during all preincubation periods was highly correlated to (i) SMBC using CFI determined on field-moist and dried soil with all preincubation periods and (ii) net N mineralization during 21 d for the Weswood soil, as well as for seven additional soil series each having five to eight levels of SOC. The CO_2-C evolved during the first day after rewetting of dried soil is recommended for rapid estimation of SMBC and potential N mineralization because of its simplicity and precision.

THE IMPORTANCE of soil microorganisms to soil fertility is recognized, but rapid, accurate soil testing procedures that reflect potential C and N mineralization have not been routinely adopted (Keeney, 1982). A valid index of soil N availability that is simple, rapid, and reproducible may preclude the use of a biological method despite its importance, because of the long time period required to estimate the relatively small amount of mineralized N due to microbial activity. Incubations lasting 1 to 2 wk for determination of mineral N accumulation are considered too time-consuming for adoption by routine soil testing programs.

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Published in Soil Sci. Soc. Am. J. 60:1133–1139 (1996).

The N-supplying potential of agricultural soils has been related to SMB and its activity (Carter and Rennie, 1982; Doran, 1987; Franzluebbers et al., 1994a). Measurement of SMB is sensitive to changes in the active fraction of SOM (Powlson et al., 1987; Anderson and Domsch, 1989) and therefore, should provide insight into the potential of soils to mineralize N. The most commonly used method for estimating SMB is CFI, although field-moist soil and a 10-d incubation are needed (Jenkinson and Ladd, 1981; Nannipieri et al., 1990; Parkinson and Coleman, 1991), which limit its adoption by soil testing programs. Several rapid methods for estimating SMB and its activity (i.e., C and N mineralization) have been developed during the past few decades including SIR (Anderson and Domsch, 1978) and AA (Alef and Kleiner, 1986), which require only 1 to 6 h of incubation, but as described, also require use of field-moist soil.

Soil testing protocol normally requires that dried soil be used because samples collected by producers and soil testing services are shipped to the soil testing facility, which may take several days, thereby altering the biochemical status if kept moist. We hypothesized that dried, rewetted, and preincubated soil could be used to obtain an estimate of SMB. The optimum preincubation period for estimating SMB and mineralizable N, therefore, needs to be established.

Our objectives were to: (i) evaluate the feasibility of using dried and preincubated soil for measurement of AA, SIR, cumulative C and net N mineralization, and SMBC using CFI and (ii) determine the optimum time of preincubation for these biochemical estimates.

MATERIALS AND METHODS

Five soil samples with SMBC levels ranging from 279 to 1260 mg kg^{-1} soil (Table 1) were collected shortly after planting wheat (*Triticum aestivum* L.) in November 1991 from a long-term field experiment established on a Weswood silty clay loam in 1982 (Table 2). Fifteen soil cores (19 mm diam.) per 4 by 12.2 m plot were collected and composited from three replications of the treatments listed in Table 1. Field-moist

Abbreviations: AA, arginine ammonification; BSR, basal soil respiration; CFI, chloroform fumigation–incubation; SIR, substrate-induced respiration; SMB, soil microbial biomass; SMBC, soil microbial biomass carbon; SOC, soil organic carbon; SOM, soil organic matter.