

# Nitrous Oxide Fluxes in Turfgrass: Effects of Nitrogen Fertilization Rates and Types

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## ABSTRACT

Urban ecosystems are rapidly expanding and their effects on atmospheric nitrous oxide (N<sub>2</sub>O) inventories are unknown. Our objectives were to: (i) measure the magnitude, seasonal patterns, and annual emissions of N<sub>2</sub>O in turfgrass; (ii) evaluate effects of fertilization with a high and low rate of urea N; and (iii) evaluate effects of urea and ammonium sulfate on N<sub>2</sub>O emissions in turfgrass. Nitrogen fertilizers were applied to turfgrass: (i) urea, high rate (UH; 250 kg N ha<sup>-1</sup> yr<sup>-1</sup>); (ii) urea, low rate (UL; 50 kg N ha<sup>-1</sup> yr<sup>-1</sup>); and (iii) ammonium sulfate, high rate (AS; 250 kg N ha<sup>-1</sup> yr<sup>-1</sup>); high N rates were applied in five split applications. Soil fluxes of N<sub>2</sub>O were measured weekly for 1 yr using static surface chambers and analyzing N<sub>2</sub>O by gas chromatography. Fluxes of N<sub>2</sub>O ranged from -22 μg N<sub>2</sub>O-N m<sup>-2</sup> h<sup>-1</sup> during winter to 407 μg N<sub>2</sub>O-N m<sup>-2</sup> h<sup>-1</sup> after fall fertilization. Nitrogen fertilization increased N<sub>2</sub>O emissions by up to 15 times within 3 d, although the amount of increase differed after each fertilization. Increases were greater when significant precipitation occurred within 3 d after fertilization. Cumulative annual emissions of N<sub>2</sub>O-N were 1.65 kg ha<sup>-1</sup> in UH, 1.60 kg ha<sup>-1</sup> in AS, and 1.01 kg ha<sup>-1</sup> in UL. Thus, annual N<sub>2</sub>O emissions increased 63% in turfgrass fertilized at the high compared with the low rate of urea, but no significant effects were observed between the two fertilizer types. Results suggest that N fertilization rates may be managed to mitigate N<sub>2</sub>O emissions in turfgrass ecosystems.

ANTHROPOGENIC ACTIVITIES have contributed to an increase in concentrations of atmospheric N<sub>2</sub>O, and agricultural soils may contribute more than 80% of anthropogenic N<sub>2</sub>O emissions (Isermann, 1994; Lal et al., 1999; Follet et al., 2000). A number of studies have determined that N<sub>2</sub>O fluxes into the atmosphere are high in croplands on which N fertilization and irrigation rates are also high (Goodroad and Keeney, 1984; Sexstone et al., 1985; Mosier et al., 1986, 1991, 1996; Bouwman, 1996). However, urbanization in the USA and elsewhere is replacing significant tracts of land with turfgrass that were once occupied by natural or agricultural ecosystems (Heimlich and Reining, 1989; Dillman et al., 1993; USGS, 1999; Alig et al., 2004; Kaye et al., 2004), and urban expansion in the USA alone is projected to increase by 79% by 2025 (Alig et al., 2004). Because turfgrass is often irrigated and fertilized with N, urban areas may represent an unappreciated, but increasingly significant contributor to atmospheric N<sub>2</sub>O.

The role of turfgrass ecosystems in atmospheric N<sub>2</sub>O inventories is unknown (Bouwman et al., 1995; Kroeze, 1999). Turfgrass has historically been excluded from re-

gional N<sub>2</sub>O inventories because of the assumption that urban land area is too small to make significant contributions. However, recent studies have indicated that 16 to 20 million ha of urbanized land, or up to 10% of the land area in some regions in the United States, are covered with turfgrasses (e.g., golf courses, sports fields, parks, home lawns, etc.) (Welterlen, 1997; NASS, 2004); this represents an area three times larger than any irrigated crop (Milesi et al., 2005).

Kaye et al. (2004) determined that turfgrass emitted 10 times more N<sub>2</sub>O to the atmosphere than did native grassland and wheat-fallow soils in northeastern Colorado, USA. Those authors also estimated that turfgrass, which covered 6.4% of the land area in their 1578 km<sup>2</sup> study, contributed as much as 30% of the N<sub>2</sub>O emissions. Research is needed to quantify N<sub>2</sub>O fluxes in turfgrasses including under various forms of land management (e.g., fertilization, irrigation, turfgrass species). Obtaining these data may be a first step in the development of best management practices in turfgrass that may mitigate the greenhouse effect by reducing effluxes of N<sub>2</sub>O, which has about 300 times the warming power of CO<sub>2</sub> (IPCC, 2001).

Fluxes of as much as 7528 μg N<sub>2</sub>O-N m<sup>-2</sup> h<sup>-1</sup> have been reported in perennial ryegrass (*Lolium perenne* L.) after N fertilization (Ryden, 1981; Maggiotto et al., 2000). Maggiotto et al. (2000) determined that fertilizer type had significant effects on N<sub>2</sub>O fluxes, with urea-based fertilizers resulting in lower N<sub>2</sub>O emissions than other fertilizer types. Conversely, fertilizer type had no impact on N<sub>2</sub>O emissions from a sward of Kentucky bluegrass (*Poa pratensis* L.) (Bergstrom et al., 2001). Others have reported that soil temperatures of 30°C or higher coupled with saturated soil conditions increased denitrification rates in Kentucky bluegrass sod (Mancino et al., 1988); greater denitrification rates typically result in higher N<sub>2</sub>O emissions (Firestone and Davidson, 1989). Irrigation with as little as 5 mm of water increased emissions of N<sub>2</sub>O markedly in a closely mown, mixed-grass sward of ryegrass and paspalum (*Paspalum dilatatum* Poir.) (Denmead et al., 1979). In another study, annual emissions of N<sub>2</sub>O-N were estimated at 2.4 kg ha<sup>-1</sup> from Kentucky bluegrass (Kaye et al., 2004).

Fluxes of N<sub>2</sub>O from the soil surface originate primarily from the processes of nitrification and denitrification in the soil (Fig. 1) (Firestone and Davidson, 1989). During nitrification, aerobic microbial populations convert ammonium (NH<sub>4</sub><sup>+</sup>) to nitrate (NO<sub>3</sub><sup>-</sup>); both NH<sub>4</sub><sup>+</sup> and NO<sub>3</sub><sup>-</sup> are readily accessible by plant roots. Thereafter, NO<sub>3</sub><sup>-</sup> can be converted to N<sub>2</sub> by anaerobic microbial populations via denitrification.

In turfgrass, the amount of N<sub>2</sub>O evolution in the soil is affected by a number of cultural practices that closely

**Abbreviations:** AS, High rate of ammonium sulfate fertilization; DOY, day of year; PVC, poly-vinyl chloride; UH, High rate of urea fertilization; UL, Low rate of urea fertilization.

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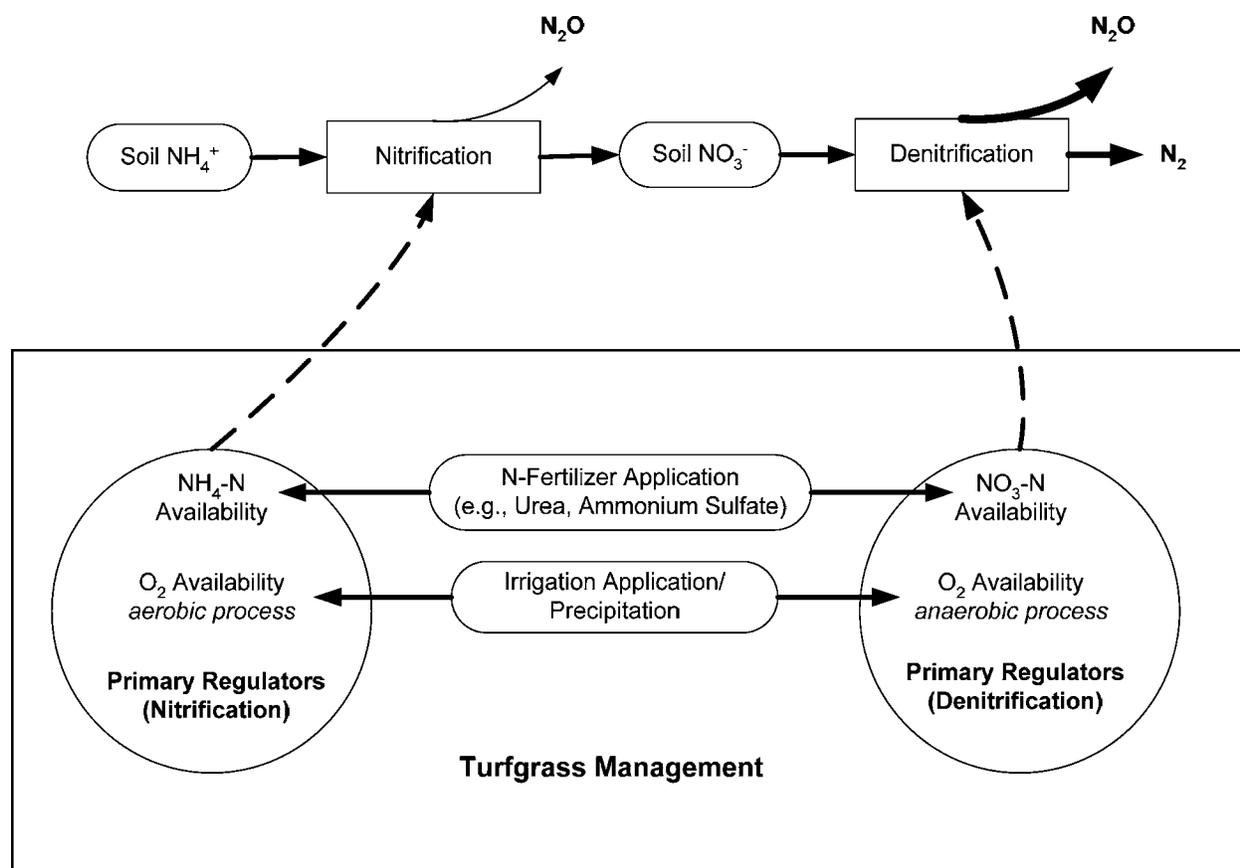


Fig. 1. Simplified conceptual model illustrating the effect of fertilization and irrigation or precipitation on  $\text{N}_2\text{O}$  emissions from the soil.

regulate nitrification and denitrification (Fig. 1). For example, ammonium-based fertilizers (e.g., ammonium sulfate) add ammonium ( $\text{NH}_4^+$ ) directly to the soil and urea fertilizers are quickly converted to  $\text{NH}_4^+$  in the soil, which is facilitated by the addition of water (e.g., from irrigation or precipitation). Nitrogen fertilization enhances nitrification, which requires  $\text{NH}_4^+$  to proceed. Nitrification also requires adequate amounts of  $\text{O}_2$  along with the presence of nitrifying microorganisms. Nitrification rates will decrease as soil water content approaches saturation (e.g., from irrigation or precipitation) because  $\text{O}_2$  availability becomes limiting. Conversely, extremely dry soils may also slow nitrification if lack of moisture restricts microbial activity.

The general requirements for denitrification to occur include the restriction of  $\text{O}_2$ , the availability of N oxides,  $\text{NO}_3^-$ ,  $\text{NO}_2^-$ ,  $\text{NO}$ , or  $\text{N}_2\text{O}$ , an available organic C source, and the presence of denitrifying microorganisms; the restriction of  $\text{O}_2$  availability is likely the dominant factor limiting denitrification in soils followed by  $\text{NO}_3^-$  availability and then available organic C (Myrold, 2004). Thus, the effects of various environmental controllers of denitrification and nitrification are somewhat hierarchical, ranging from more to less important (Firestone and Davidson, 1989; Myrold, 2004). Nitrogen fertilization and irrigation or precipitation affect both nitrification and denitrification and hence,  $\text{N}_2\text{O}$  fluxes, primarily by affecting proximal factors in the hierarchy (i.e., the amount of  $\text{NH}_4^+$ ,  $\text{NO}_3^-$ , and  $\text{O}_2$  availability in the soil).

Other factors not discussed here may also affect nitrification and denitrification rates (e.g., physical properties of soils, temperature, pH). A comprehensive discussion of these factors was not included for the sake of brevity and because this study focused on the effects of N fertilization on  $\text{N}_2\text{O}$  fluxes. More complete descriptions of these factors and how they impact nitrification and denitrification are available from a number of sources (Firestone and Davidson, 1989; Myrold, 2004).

In this study we measured  $\text{N}_2\text{O}$  emissions from turfgrass fertilized with two rates of urea and one rate of ammonium sulfate. The specific objectives of this study were to determine: (i) the magnitude, seasonal patterns, and cumulative annual emissions of  $\text{N}_2\text{O}$ -N fluxes in a turfgrass ecosystem; (ii) the effects of a high and a low N-fertilization rate of urea on  $\text{N}_2\text{O}$  fluxes; and (iii) the effects of two N-fertilizer types, urea and ammonium sulfate, on  $\text{N}_2\text{O}$  fluxes.

## MATERIALS AND METHODS

The field study was conducted from Day of Year (DOY) 276, 2003 to DOY 279, 2004 at the Rocky Ford Turfgrass Research Center near Manhattan, Kansas (Rocky Ford; 39.12°N, 96.35°W). The soil at the site was a Chase silt loam (fine, montmorillonitic, mesic, Aquic, Argiudolls). Mean annual air temperature was 11.6°C, which was 1.1°C below the 30-yr mean; precipitation was 887 mm, or 3 mm above normal. The summer months were substantially wetter and cooler than normal; precipitation between June and August 2004 was 468 mm,

which was 148 mm above normal, and the average temperature was 23.2°C, or 2.1°C below normal for the same period.

### Fertilizer Rate and Type

Twenty four plots (2 by 2 m each) were established in September 2003 in an existing sward of perennial ryegrass; plots were separated by a minimum of 2 m. Three fertilizer treatments, replicated 8 times each, were applied to plots in a randomized block design. The treatments represented typical N fertilizer amounts and types in turfgrass and included: (i) urea N at a high rate (UH; 250 kg N ha<sup>-1</sup> yr<sup>-1</sup>); (ii) urea N at a low rate (UL; 50 kg N ha<sup>-1</sup> yr<sup>-1</sup>); and (iii) ammonium sulfate N at a high rate (AS; 250 kg N ha<sup>-1</sup> yr<sup>-1</sup>). The high N rate is representative of intensively managed turfgrass such as in golf courses, and the low rate a minimally maintained homeowner lawn. In UH and AS, urea and ammonium sulfate were applied in split applications of: 75 kg N ha<sup>-1</sup> on DOY 279, 2003; 50 kg N ha<sup>-1</sup> on DOY 318, 2003, DOY 86, 2004, and DOY 128, 2004; and 25 kg N ha<sup>-1</sup> on DOY 191, 2004. In UL, 50 kg N ha<sup>-1</sup> was applied on DOY 279, 2003.

After fertilizations, plots were irrigated with about 15 mm of water to incorporate fertilizer into the soil and reduce ammonia volatilization (Bowman et al., 1987). Plots were irrigated one to three times weekly or as needed to prevent drought stress. Irrigation applications were measured with in-line flow meters (Model FTB6205, Omega Engineering, Stamford, CT). Plots were mowed once or twice weekly as needed at a height of 7.5 cm with a walk-behind rotary mower.

The plot area was treated with herbicides on DOY 295, 2003 and DOY 148, 2004 for broadleaf control (carfentrazone-ethyl; 2,4-D ethylhexyl ester; Mecoprop-p acid; Dicamba, acid; 4.68 L ha<sup>-1</sup>) and on DOY 148, 2004 for control of grassy weeds (dithiopyr; 4.68 L ha<sup>-1</sup>). Fungicide (chlorothalonil; 14.00 L ha<sup>-1</sup>) was applied on DOY 149, 2004 for dollarspot control, and an insecticide (halofenozide; 112.3 kg ha<sup>-1</sup>) was applied on DOY 191, 2004 for the control of white grubs.

### Measurements of Nitrous Oxide Fluxes

Soil-surface N<sub>2</sub>O fluxes were measured by placing static, vented poly-vinyl chloride (PVC) chambers (7.5 cm high by 20-cm diam.) using the method described by Hutchinson and Mosier (1981; Mosier et al., 1991, 1997; Kaye et al., 2004). Using this method, temperatures inside the chambers were assumed to be equivalent with ambient air temperatures measured at a nearby weather station. Ambient temperatures inside the chambers were preserved by constructing them with white PVC and by placing highly reflective aluminum foil tape on the outside of the chambers to reflect radiation away from the chamber. In this study, gas was not mixed mechanically inside the chambers. In general, thermal buoyancy and/or external turbulence-induced pressure fluctuations support uniform mixing of the chamber head space and furthermore, mechanical gas mixing may be undesirable because it may artificially enhance gas exchange rates (Hutchinson and Livingston, 2002).

Permanent PVC collars were placed randomly at one location in each plot and were driven approximately 8 cm into the soil. On measurement days, chambers were installed on the collars and gas samples from inside the chambers were removed with 12-mL polypropylene syringes fitted with nylon stopcocks at 0, 30, and 60 min. Gas samples were transported to the lab and analyzed by a gas chromatograph (Shimadzu GC14B, Shimadzu Scientific Instruments, Columbia, MD) equipped with a Porapak Q column (3.175 × 10<sup>-3</sup>-m diam. × 1 m, 80/100 mesh) and an electron capture detector. Gas samples were always analyzed on the same day as collected, and

generally within 6 h. Fluxes were calculated as described by Hutchinson and Mosier (1981) and Mosier et al. (1991).

The sampling frequency in the field was generally once weekly although measurements were more frequent after fertilizations to capture transient peaks in N<sub>2</sub>O fluxes (Christensen, 1983; Sexstone et al., 1985). Fluxes were not sampled during one 35-d period in late January through mid February (DOY 20 to 55, 2004) because snow cover prevented the measurement of fluxes. Gas samples were usually collected from 0700 to 1100 CST on each measurement day to reduce the impact of diurnal variations in N<sub>2</sub>O emissions (Blackmer et al., 1982; Christensen, 1983; Goodroad and Keeney, 1984).

### Ancillary Measurements

Soil properties, soil temperature, volumetric soil water content, and periodically, NH<sub>4</sub><sup>+</sup>, NO<sub>3</sub><sup>-</sup>, and aboveground biomass production were measured to evaluate their relationships with N<sub>2</sub>O emissions. Soil properties, including pH, texture, and bulk density, were measured in the 0- to 10-, 10- to 20-, and 20- to 30-cm profiles (Table 1). Ammonium and nitrate concentrations in the 0- to 10-cm profile were measured intensely on 4 d during the growing season, concurrent with N<sub>2</sub>O flux measurements (DOY 147, 174, 202, and 216, 2004). On each of the 4 d, soils from each of the 36 plots were sampled separately for NH<sub>4</sub><sup>+</sup> and NO<sub>3</sub><sup>-</sup> concentration to provide rigorous, direct correlations with N<sub>2</sub>O fluxes from each respective plot.

Climatological variables were measured at a weather station located at Rocky Ford Turfgrass Research Center. To determine irrigation requirements, evapotranspiration (ET) was calculated by using the Penman-Monteith equation (FAO-56; FAO, 1998).

Soil water filled porosity (WFP) and temperature at 5 cm were measured automatically using the DPHP technique (Campbell et al., 1991; Tarara and Ham, 1997; Song et al., 1998). Sensors were fabricated in the laboratory as described by Basinger et al. (2003) and Bremer (2003). Measurements of WFP were logged once daily at 0600 CST and soil temperatures were logged every 60 min. All data acquisition and control were accomplished with a micrologger and accessories (CR10x and one AM16/32, Campbell Scientific, Logan, UT). Bulk density of the soil at 5 cm was 1.35 g cm<sup>-3</sup> (determined from volumetric samples 5.4-cm diam. by 3 cm); organic matter was 4% (Soil Testing Laboratory, Kansas State University).

Clippings were collected periodically from DOY 159 to 288, 2004 with a walk-behind rotary mower equipped with a modified collection bag that allowed for complete capture of clippings from each plot. Clipped biomass was determined gravimetrically after samples had been dried in a forced-air oven for 48 h at 65°C.

### Data Analysis

Nitrous oxide flux data from each plot on each measurement day were evaluated to ensure fitness, and flux rates were then calculated. Cumulative emissions of N<sub>2</sub>O were estimated as the sum of the products of weekly flux rates and the number of days between samples. Cumulative values were first calculated for each plot, and treatment values were then obtained

**Table 1. Soil site properties in study area.**

Depth cm	pH	Bulk density g cm <sup>-3</sup>	Sand			Clay			Silt		
			%			%			%		
0 to 10	6.6	1.35	14	29	57						
10 to 20	7.2	1.42	14	31	55						
20 to 30	7.2	1.47	14	34	52						

by averaging cumulative values over all plots within each respective treatment. If a measurement preceded or followed fertilization, which had large effects on  $N_2O$  emissions, the interpolation was truncated based on the timing of the fertilization.

Tests of differences among treatments in measurements of  $N_2O$  fluxes, gravimetric estimates of clipping biomass, and ammonium and nitrate concentrations in the soil were conducted by using the mixed linear model of SAS ( $P < 0.05$ ; SAS Institute, 2003); fixed effects included fertilizer type and amount and random effects included block. Correlations between soil N and  $N_2O$  fluxes were conducted with the correlation procedure (Spearman) of SAS.

## RESULTS AND DISCUSSION

### Fluxes from Fall 2003 through Spring 2004

Fluxes of  $N_2O$ -N increased by 5 to 15 times in UH and AS plots within 3 d after fertilization in the fall of 2003 (Fig. 2) and were the highest observed in the year-long study; 36 mm of rainfall 2 d after fertilization likely contributed to the higher fluxes (Sexstone et al., 1985; Clayton et al., 1994). Peak fluxes were measured 3 d after fertilization at  $407 \mu\text{g } N_2O\text{-N m}^{-2} \text{ h}^{-1}$  in UH and were comparable to  $N_2O$ -N fluxes measured in fertilized crops and grasslands (Mosier et al., 1991; Clayton et al., 1994; Kaiser et al., 1998; Webb et al., 2004). Increases in  $N_2O$  fluxes are not unusual after fertilization in crops or grasslands (Mosier et al., 1991; Clayton et al., 1994; Webb et al., 2004) although no change or even negative fluxes have been observed after N fertilizations (Maggiotto et al., 2000; Webb et al., 2004).

Fluxes decreased rapidly after peaking and generally began to level off after about 1 to 2 wk, which is similar to the duration of increases in  $N_2O$  emissions observed after N fertilization in other grassland or turf studies (Ryden, 1981; Christensen, 1983; Bergstrom et al., 2001). Fluxes were significantly higher in UH than in AS (DOY 280, 2003) and UL (DOY 283, 2003) (Fig. 2), and cumu-

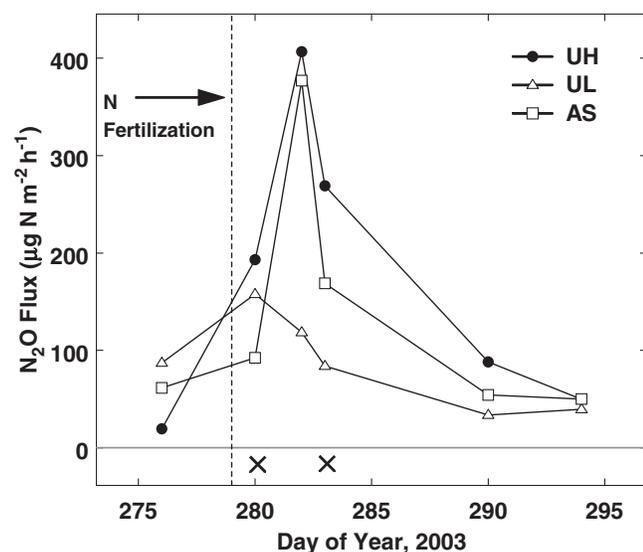


Fig. 2. Fluxes of N ( $N_2O$ -N) from perennial ryegrass in the fall of 2003. Vertical dashed lines represent N-fertilization date. Symbols (x) along the abscissa indicate significant differences between at least 2 treatments ( $P < 0.05$ ).

lative fluxes in the week following fertilization were significantly higher in UH than in UL (data not shown).

Later in the fall of 2003, fluxes again increased in UH and AS after N fertilization in those plots (Fig. 3A). Four days after fertilization (DOY 322, 2003), fluxes were significantly higher in UH and AS than in UL, and peaked at 68 and  $55 \mu\text{g } N_2O\text{-N m}^{-2} \text{ h}^{-1}$  in UH and AS, respectively.

With the exception of the increase after N fertilization,  $N_2O$  fluxes generally decreased among treatments during the fall and early winter of 2003 (DOY 301 to 357, 2003), following the trend of soil and air temperatures (Fig. 3B). On DOY 346, 2003, fluxes were negative for the first time and were the largest negative fluxes observed during the study; the greatest negative fluxes were  $-22 \mu\text{g } N_2O\text{-N m}^{-2} \text{ h}^{-1}$  in UH. Lower temperatures, moist soils (Fig. 3b and 3c), and low  $NO_3^-$  availability presumably resulted in the consumption of  $N_2O$  in the soil (and hence, from the atmosphere) by denitrifying bacteria (Firestone and Davidson, 1989). Negative  $N_2O$

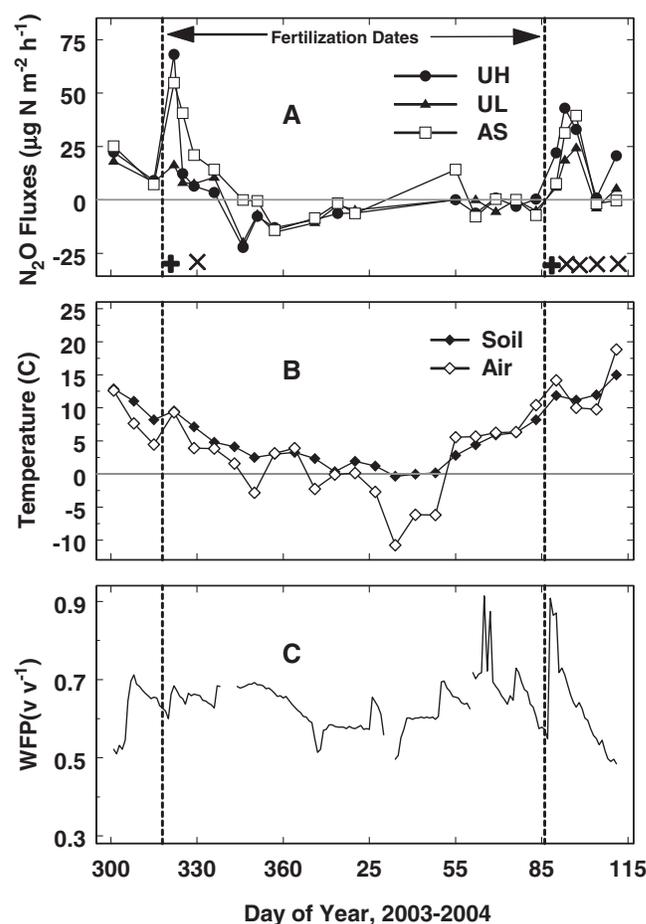


Fig. 3. (A) Fluxes of  $N_2O$ -N from perennial ryegrass; (B) average soil temperature at 5 cm among plots and air temperature at 2 m; and (C) average soil water filled porosity (WFP) at 5 cm among plots from late October 2003 to mid April 2004, Day of Year (DOY) 301, 2003 to DOY 111, 2004. Data for WFP are presented as daily and soil and air temperatures as 7-d averages. Vertical dashed lines represent fertilization dates. Symbols (x) along the abscissa in (A) indicate significant differences between at least 2 treatments ( $P < 0.05$ ) and plus (+) indicate significant differences between one and the other 2 treatments.

fluxes have been observed in a number of ecosystems (Mosier et al., 1996; Teira–Esmatges et al., 1998; Webb et al., 2004), including in turfgrass (Ryden, 1981). With few exceptions, fluxes were generally slightly negative or negligible during the winter until after the spring fertilization on DOY 86, 2004. Small winter fluxes in this study contrast with fluxes in some native grasslands and fertilized crops in which 20 to 50% of annual  $N_2O$  emissions occurred during winter (Mosier et al., 1996; Kaiser et al., 1998).

On DOY 55, 2004,  $N_2O$  fluxes were positive in AS, but differences were not statistically significant among treatments (Fig. 3). The higher average flux in AS on DOY 55 resulted from a high flux rate in only one plot, which indicates large spatial variability of  $N_2O$  emissions from the soil as reported by other researchers (Parkin, 1987; Clayton et al., 1994; Bergstrom et al., 2001).

After N fertilization in UH and AS on DOY 86, 2004,  $N_2O$  fluxes increased among plots, including in UL which were not fertilized (Fig. 3A). The increase in UL was likely caused by an increase in soil water content from 38 mm of precipitation in the 2 d after fertilization (Fig. 3C) (Denmead et al., 1979; Sexstone et al., 1985). Precipitation also may have caused additional increases in  $N_2O$  fluxes in UH and AS and thus, amplified or confounded the effects of N fertilization (Clayton et al., 1994). Fluxes of  $N_2O$  were generally higher in fertilized than in unfertilized plots during a 25-d period after fertilization, although fluxes in AS were significantly lower than in UH on 2 of the 5 measurement dates during that period. Significant differences remained among treatments on DOY 111, 2004, which was more than 3 wk after N fertilization.

### Fluxes during Summer 2004

During the summer of 2004,  $N_2O$  fluxes again increased markedly after N fertilization on DOY 128 (Fig. 4A). Four days after fertilization, fluxes of  $N_2O$  had increased in all treatments including in UL, which was not fertilized at that time. As in the earlier situation (after N fertilization on DOY 86, 2004, Fig. 3), 24 mm of precipitation 3 d after fertilization (DOY 131, 2004) increased soil moisture (Fig. 4C) and thus, likely inflated fluxes in UL. In UH and AS, precipitation also may have amplified or confounded the effects of fertilization on  $N_2O$  fluxes. Soil temperatures also increased by nearly 5°C between the pre- and post-fertilization measurements (Fig. 4B) and may have further inflated  $N_2O$  fluxes among treatments (Denmead et al., 1979; Christensen, 1983; Goodroad and Keeney, 1984).

Later, after fertilization of UH and AS on DOY 191, 2004,  $N_2O$  fluxes increased considerably in UH and AS but only slightly in UL. The slight increase in UL, despite receiving no N fertilization, was similar to a small increase observed in UL on DOY 318, 2003 (Fig. 3A), after fertilization of only UH and AS. On neither of these occasions was precipitation a factor. However, post-fertilization irrigations that were applied to all plots, including UL, may have contributed to the slight increases in  $N_2O$  emissions in UL (Denmead et al., 1979; Sexstone et al., 1985).

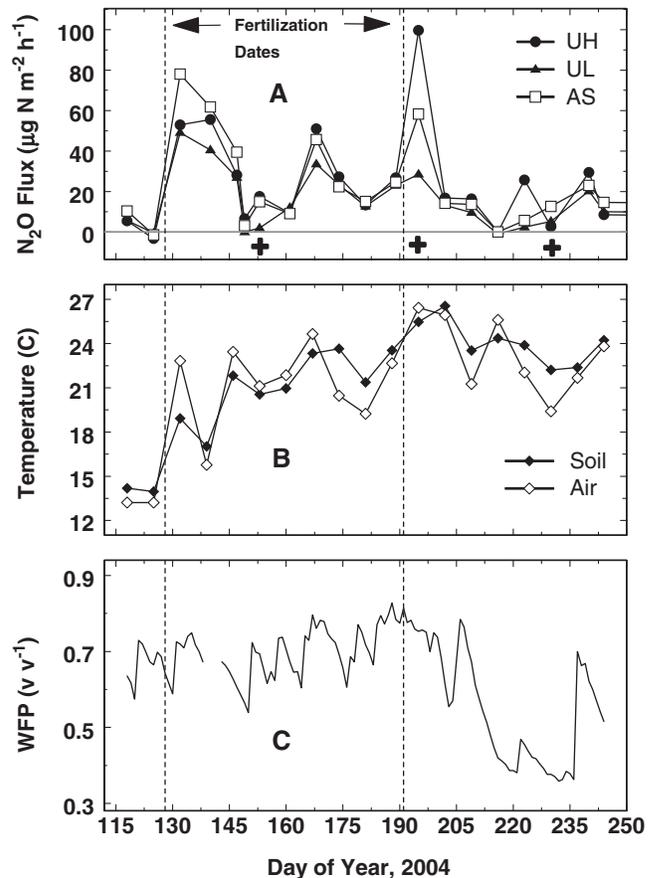


Fig. 4. (A) Fluxes of  $N_2O$ -N from perennial ryegrass; (B) average soil temperature at 5 cm among plots and air temperature at 2 m; and (C) average soil water filled porosity (WFP) at 5 cm among plots. Daily averages are presented for WFP and 7-d averages for soil and air temperatures. Vertical dashed lines represent fertilization dates. Symbols plus (+) along the abscissa in (A) indicate significant differences ( $P < 0.05$ ) between one and the other 2 treatments.

Correlations were significant but weak between  $N_2O$  fluxes and soil  $NO_3^-$  ( $r = 0.52$ ;  $P < .0001$ ) and non-significant between  $N_2O$  fluxes and  $NH_4^+$  (data not shown). Low correlations between  $N_2O$  fluxes and soil  $NH_4^+$  and  $NO_3^-$  have been reported by others and some have suggested that  $N_2O$  emissions may be more related to soil N turnover rates than to mineral N pool size (Schimel et al., 1989; Davidson and Hackler, 1994; Matson and Vitousek, 1990). Mosier et al. (1996) found low correlations between  $N_2O$  fluxes and soil N concentrations and reported that because  $N_2O$  production is determined by complex interactions among soil N concentration, water, temperature, etc., direct correlations between  $N_2O$  fluxes and any one of these variables was typically low.

Aboveground biomass production (clippings), which indicates the availability of organic C that is an important controller of denitrification, also was not a strong indicator of  $N_2O$  fluxes (data not shown). Clippings were significantly less in UL than in UH and AS on all measurement dates except for the first (DOY 148, 2004), and yet  $N_2O$  fluxes were only significantly lower in UL in three out of 15 d when fluxes were measured between DOY 118 and 244, 2004.

## Annual Fluxes of Nitrous Oxide and Comparisons to Other Ecosystems

Cumulative annual emissions of  $\text{N}_2\text{O}$ -N were 58 and 63% greater in UH and AS, respectively, than in UL (Fig. 5). Cumulative emissions of  $\text{N}_2\text{O}$ -N were  $1.65 \text{ kg ha}^{-1} \text{ yr}^{-1}$  in UH,  $1.60 \text{ kg ha}^{-1} \text{ yr}^{-1}$  in AS, and  $1.01 \text{ kg ha}^{-1} \text{ yr}^{-1}$  in UL; annual  $\text{N}_2\text{O}$  emissions were significantly greater from UH than UL. Thus, although transient differences between weekly fluxes in UH and AS were observed during the study, differences in annual  $\text{N}_2\text{O}$  emissions between UH and AS were not significant, indicating fertilizer type had no effect on  $\text{N}_2\text{O}$  emissions in this study.

Over the year, an equivalent of approximately 0.65% in UH and AS and 2.02% in UL of annually applied N fertilizer was emitted into the atmosphere during the 1-yr study. These results are similar to the 2.2% of added N fertilizer emitted into the atmosphere as reported in the only other experiment where annual  $\text{N}_2\text{O}$  emissions were measured in turfgrass ecosystems (Kaye et al., 2004). In the latter experiment, annual  $\text{N}_2\text{O}$  emissions in turfgrass were comparable to emissions in irrigated corn, and were 10 times greater than in native grassland and wheat-fallow soils.

Cumulative emissions of  $\text{N}_2\text{O}$  were rapid during high fluxes after the fall fertilization in 2003 (Fig. 5). However, during the winter when soils were cool and wet and  $\text{N}_2\text{O}$  fluxes were slightly negative (Fig. 3a), cumulative  $\text{N}_2\text{O}$  emissions decreased as soil microbes absorbed atmospheric  $\text{N}_2\text{O}$  during denitrification. Reduction of  $\text{N}_2\text{O}$  to  $\text{N}_2$  is possible when the production of  $\text{N}_2\text{O}$  is limited by a low supply of  $\text{NO}_3^-$ . As a result, denitrifiers use  $\text{N}_2\text{O}$  as the substrate. (Dr. Charles Rice, personal communication, 2005). During the winter period, cumulative fluxes of  $\text{N}_2\text{O}$  decreased by 0.11 to  $0.20 \text{ kg ha}^{-1}$  among treatments compared with cumulative emissions during October and November. Thereafter, during the

spring and summer months, cumulative  $\text{N}_2\text{O}$  emissions increased because fluxes were generally positive.

In general, annual emissions of  $\text{N}_2\text{O}$  in this study were similar to or lower than  $\text{N}_2\text{O}$  fluxes from other urban or agricultural grasslands. For example, annual emissions from a Kentucky bluegrass lawn were  $2.4 \text{ kg N}_2\text{O-N ha}^{-1} \text{ yr}^{-1}$  during a 1-yr study (Kaye et al., 2004), and fluxes from native grassland and wheat ecosystems in the USA ranged from  $0.24$  to  $2.26 \text{ kg N}_2\text{O-N ha}^{-1} \text{ yr}^{-1}$  (Mosier et al., 1991, 1996). Dobbie et al. (1999) reported annual emissions in cut ryegrass in Scotland that ranged from  $1.9$  to  $18.4 \text{ kg N}_2\text{O-N ha}^{-1} \text{ yr}^{-1}$  over a 3-yr period. In their study, significant interannual variation in  $\text{N}_2\text{O}$  fluxes resulted from the same field; because the field was not irrigated, wet and dry periods were responsible for most of the variation. Emissions of  $\text{N}_2\text{O}$  ranged from  $1.5$  to  $7.8 \text{ kg N}_2\text{O-N ha}^{-1} \text{ yr}^{-1}$  in grass, barley, and fallow in agricultural mineral soils in the boreal region (Syvasalo et al., 2004) and were  $11.2 \text{ kg N}_2\text{O-N ha}^{-1} \text{ yr}^{-1}$  in a fertilized grassland in Germany where N turnover rates were rapid (Tilsner et al., 2003). Annual emissions as high as  $56 \text{ kg N}_2\text{O-N ha}^{-1}$  were reported from rye cropland in peat soils in Germany (Flessa et al., 1998) and from long-term, manured agricultural soils in Canada (Chang et al., 1998) where pH was low and groundwater levels were high.

Estimates of annual  $\text{N}_2\text{O}$  emissions in this study also were comparable to annual  $\text{N}_2\text{O}$  emissions from a number of cereal crops, sugarbeets (*Beta vulgaris* L.), potatoes (*Solanum tuberosum*), and linseed (*Linum usitatissimum*), which never exceeded  $2 \text{ kg N}_2\text{O-N ha}^{-1} \text{ yr}^{-1}$  in a 6-yr study in the United Kingdom (Webb et al., 2004). However, emissions in the current study were lower than fluxes in rice (*Oryza sativa* L.), cotton (*Gossypium hirsutum* L.), and vegetable croplands in the United States ( $6.5$  to  $8.4 \text{ kg N}_2\text{O-N ha}^{-1} \text{ yr}^{-1}$ ; Mummey et al., 1998), from maize (*Zea mays* L.) in Germany ( $3.9$  to  $8.7 \text{ kg N}_2\text{O-N ha}^{-1} \text{ yr}^{-1}$ ; Sehy et al., 2003), and from onion (*Allium cepa* L.) cropland in Japan ( $15.6 \text{ kg N}_2\text{O-N ha}^{-1}$  during the growing season; Kusa et al., 2002) where soil water content was high and N-fertilizer was applied at a higher rate than this study. Annual emissions in the current study were also lower than in fallow agricultural organic soils in the boreal region ( $23.5 \text{ kg N}_2\text{O-N ha}^{-1} \text{ yr}^{-1}$ ; Maljanen et al., 2004); high fluxes in the latter study were from wet, N-fertilized, bare soils where the absence of plants resulted in higher soil  $\text{NO}_3^-$  concentrations.

## CONCLUSIONS

Nitrogen fertilization increased  $\text{N}_2\text{O}$  emissions by as much as 15 times within 3 d although the amount of increase varied after each fertilization; increases tended to be greater when significant precipitation occurred within 2 or 3 d of N fertilization. Fluxes of  $\text{N}_2\text{O}$  ranged from  $-22 \mu\text{g N}_2\text{O-N m}^{-2} \text{ h}^{-1}$  during winter to  $407 \mu\text{g N}_2\text{O-N m}^{-2} \text{ h}^{-1}$  after fall fertilization. Cumulative annual emissions of  $\text{N}_2\text{O}$ -N were  $1.65 \text{ kg ha}^{-1}$  in UH,  $1.60 \text{ kg ha}^{-1}$  in AS, and  $1.01 \text{ kg ha}^{-1}$  in UL. Thus, N fertilization at the high rate of urea increased annual

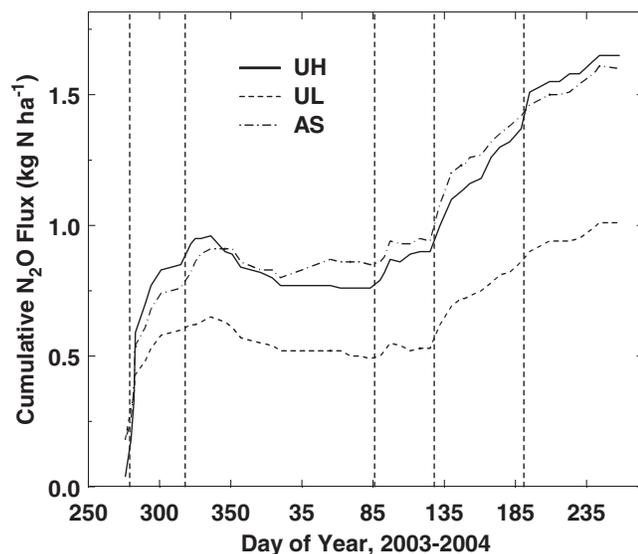


Fig. 5. Cumulative fluxes of  $\text{N}_2\text{O}$ -N from plots treated with high rates of urea (UH), low rates of urea (UL), and high rates of ammonium sulfate (AS). Vertical dashed lines represent fertilization dates.

N<sub>2</sub>O emissions in turfgrass by 63% compared with N fertilization at the low rate. Emissions of N<sub>2</sub>O were similar, however, between turfgrasses fertilized with the same rate of ammonium sulfate and urea N. In general, N<sub>2</sub>O fluxes were higher for about 2 or 3 wk after N fertilization before decreasing to background levels. Further long-term research is necessary to measure N<sub>2</sub>O emissions in turfgrasses, including in different geographical areas with different turfgrass species or considering interannual variations in climatic conditions at individual sites, and to determine best management practices that may reduce N<sub>2</sub>O emissions in turfgrasses.

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