Denitrification in Suburban Lawn Soils

Steve M. Raciti,* Amy J. Burgin, Peter M. Groffman, David N. Lewis, and Timothy J. Fahey

There is great uncertainty about the fate of nitrogen (N) added to urban and suburban lawns. We used direct flux and in situ chamber methods to measure N_2 and N_2O fluxes from lawns instrumented with soil O_2 sensors. We hypothesized that soil O_2, moisture, and available NO_3^- were the most important controls on denitrification and that N_2 and N_2O fluxes would be high following fertilizer addition and precipitation events. While our results support these hypotheses, the thresholds of soil O_2, moisture, and NO_3^- availability required to see significant N_2 fluxes were greater than expected. Denitrification rates were high in saturated, fertilized soils, but low under all other conditions. Annual denitrification was calculated to be 14.0 ± 3.6 kg N ha^{-1} yr^{-1}, with 5% of the growing season accounting for >80% of the annual activity. Denitrification is thus an important means of removing reactive N in residential landscapes, but varies markedly in space, time, and with factors that affect soil saturation (texture, structure, compaction) and NO_3^- availability (fertilization). Rates of in situ N_2O flux were low; however, when recently fertilized soils saturated with water were incubated in the laboratory, we saw extraordinarily high rates of N_2O production for the first few hours of incubation, followed by rapid N_2O consumption later in the experiment. These findings indicate a lag time between accelerated N_2O production and counterbalancing increases in N_2O consumption; thus, we cannot yet conclude that lawns are an insignificant source of N_2O in our study area.

The developed land area of the United States has grown more than 400% in the past half century (Brown et al., 2005) and continues to grow rapidly (Goetz et al., 2004; Jantz et al., 2005; Brown et al., 2005). The most visible changes that accompany development are increased impervious surface area (e.g., buildings, roads, and sidewalks) and replacement of natural vegetation and agriculture fields with lawns (i.e., turfgrass). Impervious surfaces in the United States are estimated to cover an area nearly the size of Ohio (112,610 km^2; Elvidge et al., 2004), and the area in lawns is estimated to be even larger (163,812 km^2; Milesi et al., 2005).

The rapid increase in residential land area has raised concerns about pollution associated with fertilizers and pesticides, which are used for lawn establishment and maintenance (Morton et al., 1988; Gold et al., 1988, 1990; Petrovic, 1990; Milesi et al., 2005). Concern for coastal and freshwater ecosystems is particularly high, where nitrogen pollution has contributed to numerous water quality problems, including harmful algal blooms, decreased biodiversity, fisheries declines, and "dead zones" in areas such as the Gulf of Mexico and Chesapeake Bay (Carpenter et al., 1998; Kemp et al., 2005; Paerl et al., 2006; Dodds et al., 2008).

Residential systems receive higher N inputs than natural systems and may have a high capacity for N retention (Baker et al., 2001; Wollheim et al., 2005; Groffman et al., 2009). Lawns, which are the dominant vegetation cover in residential areas, may play an important role in this outcome. While lawns can have significant N losses, especially when overwatered and overfertilized (Morton et al., 1988; Petrovic, 1990; Qian et al., 2003), they also have considerable potential for organic matter accumulation and N retention (Gold et al., 1990; Qian and Follett, 2002; Kaye et al., 2005; Golubiewski, 2006; Raciti et al., 2008).

Through our studies of the Baltimore Long-Term Ecological Research (LTER) site, we have measured N inputs, outputs, and transformations in residential ecosystems with a particular focus on lawns. In a ^{15}N-tracer study we found that when a pulse of nitrate (NO_3^-) (simulating atmospheric deposition) was added to Baltimore area lawns, the NO_3^- was rapidly incorporated or retained in microbial biomass, fine roots, aboveground vegetation biomass, thatch, and soil organic matter (Raciti et al., 2008). Overall ^{15}N retention was similar, and possibly higher, than in

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Abbreviations: BES, Baltimore Ecosystem Study; N-FARM, Nitrogen-Free Atmospheric Recirculation Method; PVC, polyvinyl chloride.

forested reference sites. In a second study, we evaluated soil carbon (C) and nitrogen (N) pools in 32 residential home lawns down to 1-m depth (Raciti et al., 2011a). We found evidence for rapid accumulation of C and N in residential soils, particularly in those with a history of agricultural land use. When we studied NO_3^- production and consumption in the same lawns (Raciti et al., 2011b), we found that net nitrification and exchangeable NO_3^- were significantly higher in residential soils than in forest soils, but these measures of NO_3^- production and availability were still notably low and comparable to deciduous forest stands in other studies (Lovett et al., 2004). Finally, while N inputs to our developed watersheds were high (Law et al., 2004), measured leaching losses were a relatively small fraction of total inputs (Groffman et al., 2009). Using information gathered from these experiments, we created a mass balance for our lawn-dominated residential systems (Raciti et al., 2011b); however, a major N flux was missing from our balance—gaseous losses of N_2 from denitrification—a process that has been notoriously difficult to measure in terrestrial ecosystems (Kulkarni et al., 2008).

Direct measurements of denitrification in terrestrial ecosystems have been limited by the difficulty of measuring N_2 fluxes (see review by Groffman et al., 2006a); however, past research suggests that denitrification may be an important pathway for N loss from turfgrass systems (e.g., Horgan et al. 2002a,b). Due to the difficulty in measuring N_2 fluxes, mass balance studies have generally estimated denitrification as the difference between recovered and unrecovered fertilizer N at the end of an experiment (Horgan et al., 2002a). For instance, Miltner and colleagues (1996) recovered 64 to 81% of ^{15}N-labeled urea in the spring and fall of their experiment, suggesting that gaseous losses might be responsible for incomplete recovery. Engelsjord and colleagues (2004) and Starr and DeRoo (1981) reported a similar range of recovery rates. Horgan and colleagues (2002a,b) attempted to account for gaseous losses by periodically measuring $^{15}N\text{-}N_2O$ and $^{15}N\text{-}N_2$ fluxes from lawn and greenhouse plots, but overall recoveries (57–84%) were not much improved over previous mass balance studies. Despite less than complete recovery, Horgan and colleagues calculated sizeable N_2 and N_2O fluxes, particularly following a major rainfall event, confirming that denitrification was an important process in turfgrass systems.

In this study, we sought to complete the N budget for our residential system (Raciti et al., 2011b) using a combination of laboratory and field measurements to quantify denitrification. There is general consensus on the proximate controls of denitrification in terrestrial ecosystems (e.g., soil temperature, moisture, oxygen, organic matter, and available NO_3^-). As an anaerobic, primarily heterotrophic process, denitrification in terrestrial environments tends to occur in small areas (hotspots) and during brief periods (hot moments) of activity when these controls converge (Parkin, 1987; McClain et al., 2003). In addition to influencing the total rate of denitrification, these factors influence the relative importance of the end products of denitrification, including N_2O, which is an important greenhouse gas (Kulkarni et al., 2008). Two problems have hindered attempts to estimate meaningful rates outside the laboratory: (i) an inability to measure direct N_2 fluxes due to high atmospheric concentrations of this gas, and (ii) an inability to measure and monitor soil oxygen (O_2) concentrations in the field (Groffman et al., 2006a). Using recent advances in instrumentation (Butterbach-Bahl et al., 2002; Burgin et al., 2010), we were able to overcome these problems to measure field-relevant rates of denitrification from lawns. Our objectives were to (i) measure soil O_2 conditions in home lawns; (ii) determine relationships between soil O_2 conditions, and N_2 and N_2O fluxes, in the laboratory and field (N_2O only); (iii) evaluate the role of soil moisture and nitrate levels on these relationships; and (iv) produce estimates of annual N gas fluxes to close the N budget of our residential ecosystem. At the outset of the study, we hypothesized that denitrification (as N_2 and N_2O) would occur on an episodic basis, with the highest rates coinciding with rainfall and after fertilizer application, when soil moisture and available NO_3^- would be high and soil oxygen concentrations would be low.

Materials and Methods

Site Description

The lawns used in this study were located in the Baltimore metropolitan area in association with the Baltimore Ecosystem Study (BES; http://beslter.org), a component of the U.S. National Science Foundation LTER network, and have been described in detail elsewhere (Groffman et al., 2009). This area has a temperate climate with warm, humid summers (1220 cooling degree days), cold winters (4720 heating degree days), and mean annual precipitation of approximately 1060 mm distributed relatively evenly throughout the year (NCDC, 2009).

The two lawns (referred to as UMBC1 and South Campus) were located on the grounds of University of Maryland Baltimore County, but were not turfgrass research plots. Campus lawns were chosen because of their known, long-term management regimes, similarities in land-use history (formerly agricultural), and their representativeness of typical residential lawns. Each lawn (or plot) contained a mixture of tall fescue (*Festuca arundinacea* Schreb.) and fine fescue (*Festuca* spp.). The UMBC1 lawn, which was less actively managed, also had significant white clover (*Trifolium repens* L.) cover. UMBC1 was fertilized each spring at a rate of 98 kg N ha^{-1} using LESCO 14–14–14 (LESCO, Cleveland, OH), applied in two applications, approximately 2 wk apart. Mowing was done at 2- to 3-wk intervals (dependent on rainfall and subsequent growth) during the spring, summer, and fall seasons to a height of approximately 10 cm. South Campus was fertilized each spring and fall at 98 kg N ha^{-1} (196 kg N ha^{-1} yr^{-1} total). Fertilizer was applied over two applications each season, approximately 2 wk apart, using LESCO 18–24–12 in the spring and LESCO 25–5–10 in the fall. The lawn was mowed weekly during the growing season to a height of approximately 10 cm. Neither lawn received irrigation, and clippings were left in place. Management of the lawns was similar to moderate (UMBC1) and moderately high maintenance (South Campus) home lawns in the study area. A lawn management survey by Law et al. (2004) found that lawn fertilizer inputs in the area ranged from zero to >300 kg N ha^{-1} yr^{-1} with a mean of 83.5 kg N ha^{-1} yr^{-1} in their suburban watershed (0.13-ha mean lot size). The soils at UMBC1 and South Campus have been classified as Joppa (loamy-skeletal, siliceous, semiactive, mesic Typic Hapludult) and Brandywine

(sandy-skeletal, mixed mesic Typic Dystrudept), respectively. Soil pH was between 5.7 and 6.0 for both lawns when measured at the beginning of the study. Atmospheric N deposition in the Baltimore metropolitan area is estimated at 11.2 kg N ha^{-1} yr^{-1} (Groffman et al., 2004).

Instrumentation

In early August of 2008, each lawn was instrumented with three Apogee diffusion-head soil O_2 sensors (SO-100 series, Apogee Instruments, Logan, UT) with the diffusion heads at 7-cm depth. The sensors in each lawn were connected to a Campbell Scientific CR-800 data logger (Campbell Scientific, Logan, UT), which was programmed to measure soil O_2 hourly over the course of approximately 1 yr (with monthly interruptions to recharge and replace batteries at each installation). Both lawns have been continually monitored for soil temperature and moisture as part of the BES network of long-term study sites (Groffman et al., 2009).

Sample Collection and Processing

Soil cores were collected to a depth of 10 cm using a 5-cm-diameter slide-hammer corer (AMS Equipment Corp., American Falls, ID) from 13 May and 4 June 2009. For each experiment (described below) an equal number of soil cores were collected from each of the two lawn areas (UMBC1 and South Campus), with the exception of the field fertilization experiment (Experiment 3), where all of the soil cores came from the UMBC1 site. Soil cores were collected randomly from each lawn area. All cores were transported to the laboratory in coolers and then stored at 4°C until they could be processed (2–4 d). A subset of soil cores from each time point were used to determine soil dry weight, percent moisture, and bulk density. Soil dry weight and percent moisture were determined by the change in mass of 10 g of field-moist soil after 48 h at 105°C. Bulk density (BD) was calculated as BD = (Total Dry Mass – Rock Mass)/(Total Volume – Rock Volume) (see Boone et al., 1999).

Direct Measurement of N_2, Nitrous Oxide, and Carbon Dioxide Fluxes from Intact Lawn Soil Cores

Denitrification measurements were made using the Nitrogen-Free Atmospheric Recirculation Method (N-FARM) system (Burgin et al., 2010), which is based on those built by Swerts et al. (1995) and Butterbach-Bahl et al. (2002). Intact soil cores were encased in 10 glass chambers (each acting as an independent incubator) with gastight lids and Swagelok fittings and valves connected to a gastight flow injection system built from Swagelok connections (Swagelok, Crawford Fitting Co., Solon, OH). These were mounted in-line with Shimadzu (Kyoto, Japan) GC8A gas chromatographs with thermal conductivity (to measure N_2, CO_2, and O_2) and electron capture (to measure N_2O) detectors. The glass chambers were submerged under water in a Plexiglas box. This served as a secondary test for leaks (which would be evident from water accumulation in the chambers) and further minimized potential gas diffusion around the chamber seals. Using this system, the ambient soil atmosphere could be replaced with an N_2–free helium (He) and O_2 atmosphere using a series of gentle vacuum-flush cycles (Butterbach-Bahl et al., 2002; Burgin et al., 2010). By lowering N_2 concentration in the soil atmosphere (to below the 7.5 μmol mol^{-1} detection limit of our thermal conductivity detector), we were able to measure small fluxes of N_2 gas exchange from intact soil cores. The system allowed us to vary soil O_2 concentrations by adjusting the He:O_2 ratio used during the vacuum-flush cycle. Regular system pressure checks, empty chambers (two blanks per incubation), and incubation of killed soil cores (autoclaved twice at 125°C for 3 h over the course of 3 d) were used to control for system leaks and residual degassing of N_2 from soil pores. All soil cores were incubated at 25°C and left completely intact (including plant roots and shoots) to minimize disturbance to the soil ecosystem.

Field Fluxes

In the field, soil:atmosphere fluxes of CO_2 and N_2O were measured using 29-cm (inner diameter) polyvinyl chloride (PVC) cylinder chambers with gas sampling ports (Bowden et al., 1991; Groffman et al., 2006b). Just before sampling, these chambers were mounted on PVC base rings installed to 5-cm depth and flush with the soil surface. These low-profile base rings allowed mowing to take place as usual between sampling intervals. At 0, 10, 20, and 30 min following placement of the chamber on the base, 9-mL gas samples were collected from gas sampling ports in the center of the chamber top by syringe. Samples were transferred to evacuated glass vials which were stored at room temperature before analysis by gas chromatography with electron capture (N_2O) or thermal conductivity (CO_2) detection.

Experiments

The first experiment tested the effects of O_2 concentration on N_2 and CO_2 fluxes at field moisture (17–20% volumetric soil moisture). Using intact soil cores, N_2 and CO_2 fluxes were measured at 20, 5, and 0% O_2 concentrations in the N-FARM. Each soil O_2 concentration was tested using fresh soils (n = 4) to avoid unknown effects that might arise from subjecting soil cores to prolonged incubation. Headspace N_2 and CO_2 concentrations were measured after 2, 6, and 24 h of incubation.

Our second experiment tested the effect of soil moisture and NO_3^- availability on N_2 and CO_2 fluxes at 0% O_2. Wet soil cores (>30% volumetric soil moisture, n = 4) were collected during a rain event and compared with dry soil cores (18% soil moisture, n = 4) collected before the rain event. Since similarly low N_2 fluxes were obtained for both conditions, a third incubation was conducted at 0% O_2 after injecting 10 mL of NO_3^- solution (approximately 100 mg N kg^{-1} dry soil) into four wet soil cores. Headspace N_2, N_2O, and CO_2 concentrations were measured after 2, 6, and 24 h of incubation.

Our third experiment examined changes in soil N_2, N_2O, and CO_2 fluxes following a field fertilization experiment. On 13 May 2009, a 3- by 30-m section of the UMBC1 site was amended with LESCO 24–0–11 fertilizer at a rate of 49 kg N ha^{-1} using a calibrated spreader. Soil cores were collected from the recently fertilized and "ambient" (i.e., control) areas of the plot. The first N-FARM incubation was performed at 20% O_2 using ambient (n = 4) and recently fertilized (n = 4) soil cores taken 24 h after fertilizer addition. The incubation was performed again at 2% soil O_2 using fresh cores. The experiment was repeated for cores taken 9 d after fertilizer addition. The fertilized area of UMBC1 included three gas rings that were used to measure N_2O and

CO_2 fluxes in the field. Field fluxes were measured before fertilizer application, and 24 h and 9 d after application (i.e., at the same time that soil cores were collected). All soil cores were incubated at field moisture (18–21% volumetric soil moisture).

We performed two more N-FARM incubations to test the effects of soil saturation (e.g., from a major rainstorm) on denitrification rates in fertilized soils. Field fertilized soil cores (collected 9 d postfertilization) were saturated by slowly dripping water onto them until no further water could be absorbed. These cores were incubated at 20 and 2% O_2 (n = 8 for each O_2 concentration).

Calculations and Statistics

Linear regression (accumulation of N_2, N_2O, or CO_2 vs. time) was used to calculate flux rates from N-FARM incubations. We used killed soil cores to correct for low levels of N_2 accumulation that might have originated from degassing from soil pores. Fluxes were expressed on a per-area basis by dividing by the ground surface area of each soil core. Field fluxes (using PVC cylinder chambers described earlier) were calculated from the linear rate of change in gas concentration, the chamber internal volume and soil surface area (Groffman et al., 2006b, 2009). Statistical differences between experimental treatments were tested using analysis of variance followed by Tukey's Honestly Significant Difference post hoc test. All statistical analyses were performed using SAS JMP version 8 statistical software (SAS Institute, 2009). For laboratory Experiments 1 and 2, an equal number of soil cores from each lawn were used on the grounds that the different management regimes would result in differences in N_2, N_2O, and CO_2 fluxes. We found no statistically significant differences between the two lawns across laboratory experiments, so the results for both lawns are grouped in the figures and statistics. Using a combination of laboratory and field measurements, we estimated annual N_2 fluxes in residential lawns (Table 1). Annual N_2 fluxes were estimated by apportioning flux rates from appropriate N-FARM incubations to the proportion of the growing season (231 d for Baltimore, MD [NCDC, 2009]) when field-measured soil moisture and O_2 conditions fell within that range.

Results

We measured in situ soil O_2 concentrations hourly for 1 yr and found that concentrations stayed within a relatively small range (Fig. 1A and 1B) near atmospheric concentrations (20%), occasionally dropping as low as 17%. A 3-wk period with two major storm systems (4.3 and 10.4 cm of precipitation) failed to cause a significant drop in bulk soil O_2 concentrations at our sites.

In laboratory incubations we tested for the effect of soil O_2 concentration on N_2 and CO_2 fluxes (at 20% field moisture), and found that CO_2 fluxes (i.e., respiration) decreased as O_2 concentrations were lowered ($p < 0.02$ for all differences). The N_2 fluxes increased as O_2 decreased, but the differences were not statistically significant due to high variability within treatments (Fig. 2A and 2B). The N_2 fluxes were low across the O_2 concentration gradient, with means ranging from 4.26 ± 3.91 μg N m^{-2} h^{-1} at 20% O_2 to 28.9 ± 17.0 μg N m^{-2} h^{-1} at 0% O_2.

We used changes in the field moisture of lawn soils (collected before and during a rain event) to measure the influence of soil moisture on N_2 and CO_2 fluxes at 0% soil O_2. While wet soils (>30% field moisture) had higher arithmetic means than dry soils (18% field moisture), these differences were not statistically significant (59.8 ± 13.9 μg N m^{-2} h^{-1} vs. 28.9 ± 17.0 μg N m^{-2} h^{-1} and 9.1 ± 1.5 mg C m^{-2} h^{-1} vs. 6.0 ± 1.8 mg C m^{-2} h^{-1}; Fig. 3A and 3B). However, when a pulse of NO_3^- was added to the wet soils incubated at 0% O_2, N_2 fluxes increased dramatically from 59.8 ± 13.9 μg N m^{-2} h^{-1} to 2216 ± 509 μg N m^{-2} h^{-1} ($p < 0.001$). Carbon dioxide fluxes from wet soils amended with NO_3^- were significantly higher than CO_2 fluxes from dry soils, but not significantly higher than fluxes in unamended wet soil cores (Fig. 3B).

Fertilized and unfertilized ("ambient") soil cores collected 24 h after fertilization in the field (applied at 49 kg N ha^{-1}) did not show significantly different N_2 and CO_2 fluxes when incubated at 20 and 2% O_2 (Fig. 4A and 4B). When this experiment was repeated with soil cores collected 9 d after fertilization (Fig. 4A), we found that soils incubated at 2% O_2 had higher denitrification rates than those incubated at 20% soil O_2; however, differences between fertilized and ambient soil cores were not statistically different at a given O_2 concentration. We found a predictable pattern of lower CO_2 fluxes at lower O_2 concentrations (Fig. 4B), but no differences in CO_2 fluxes between ambient and fertilized soils at a given O_2 concentration. Soil moisture was approximately 20% at both sampling intervals.

When we saturated field-fertilized soil cores (to simulate soil conditions after a major storm event), we found tremendously high N_2 fluxes at both 20 and 2% soil O_2 (Fig. 5A). The N_2 fluxes in saturated soils were 4039 ± 723 μg N m^{-2} h^{-1} and 2959 ± 254 μg N m^{-2} h^{-1} at 20% and 2% O_2, respectively, compared to 53.3 ± 30 μg N m^{-2} h^{-1} and 243 ± 98 μg N m^{-2} h^{-1} in field-moist soils. Carbon dioxide fluxes were lower at 2% O_2 than at 20% O_2 at each moisture level (Fig. 5B).

Field-measured fluxes of N_2O and CO_2 showed no significant differences 24 h after fertilization (Fig. 6A and 6B), but after 9 d fluxes had increased significantly compared to prefertilization rates (2.00 ± 0.62 μg N_2O-N m^{-2} h^{-1} and 43.4 ± 5.5 mg C m^{-2} h^{-1} vs. −0.07 ± 0.33 μg N m^{-2} h^{-1} and 23.0 ± 3.2 mg C m^{-2} h^{-1}).

While we were able to measure N_2O concentrations in the incubation chambers, calculating N_2O fluxes would have been misleading due to the rapid and dynamic swings in concentration observed between sampling intervals. For instance, when fertilized soils were saturated with water (or when an NO_3^- solution

Table 1. Annual denitrification rates calculated for study lawns based on laboratory measured N_2 flux rates and field measured soil moisture conditions, University of Maryland Baltimore County.

Volumetric soil moisture	Proportion of growing season	N^2 flux	Notes
		kg N ha^{-1} yr^{-1}	
8–35%	0.95	2.95 ± 1.65	Fertilized lawns 18–20% O_2, pH = 5.7–6.0
35%+	0.05	223.9 ± 40.1	Rate for saturated soils
Annual N_2 flux		14.0 ± 3.6	

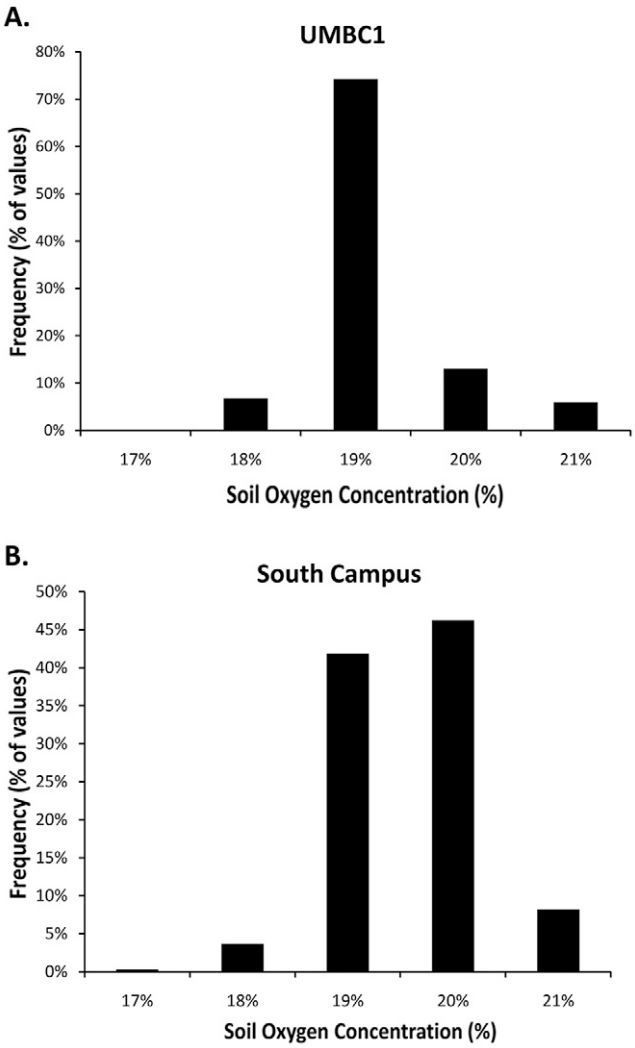

Fig. 1. Macroscale soil oxygen concentrations in two study lawns over a 1-yr time period: (A) UMBC1, and (B) South Campus, University of Maryland Baltimore County. Values for each lawn represent the mean of three sensors at 7-cm depth.

was added to wet soils), we saw extraordinarily high concentrations of N_2O for the first few hours of the incubation, followed by rapid N_2O consumption. In some cases, virtually all of the accumulated N_2O had been consumed before the next sampling interval (see Fig. 7 for examples from individual soil cores).

We estimated annual N_2 fluxes in residential lawns by apportioning flux rates from appropriate N-FARM incubations to the proportion of the growing season when field-measured soil moisture and O_2 conditions fell within that range (Table 1). We measured mean N_2 fluxes equivalent to 2.95 ± 1.65 kg N ha^{-1} yr^{-1} in UMBC1 and South Campus soils at 20% soil O_2 and 18 to 30% soil moisture. Based on soil O_2 and moisture measurements at the sites, similar conditions were found during 95% of the year, so a rate of 2.95 ± 1.65 kg N ha^{-1} yr^{-1} was assumed for this time period annually. Conversely, we measured very high N_2 flux rates (equivalent to 224 ± 40.1 kg N ha^{-1} yr^{-1}) at 20% O_2 in saturated, fertilized lawn soils. Based on maximum soil moisture readings at the sites (38%), we estimated that soils at 35% and greater soil water content would be approaching saturated conditions. Saturated conditions were experienced for 5% of the average growing season (based on 5 yr of soil moisture data). By this method, total annual denitrification was estimated at 14.0 ± 3.6 kg N ha^{-1} yr^{-1}.

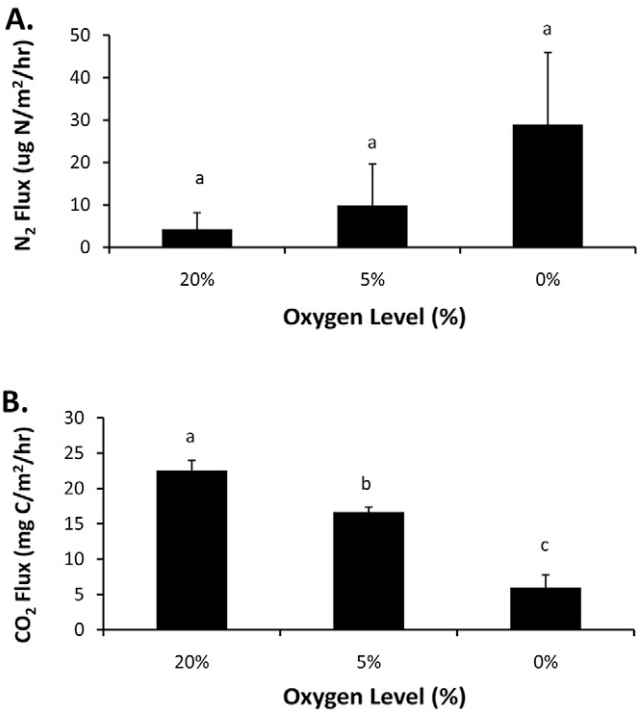

Fig. 2. (A) N_2 and (B) CO_2 fluxes from intact lawn soil cores at approximately 20% volumetric soil moisture across three soil oxygen concentrations (20, 5, and 0%), University of Maryland Baltimore County. Different letters indicate significant differences across treatments ($p < 0.05$, $n = 4$).

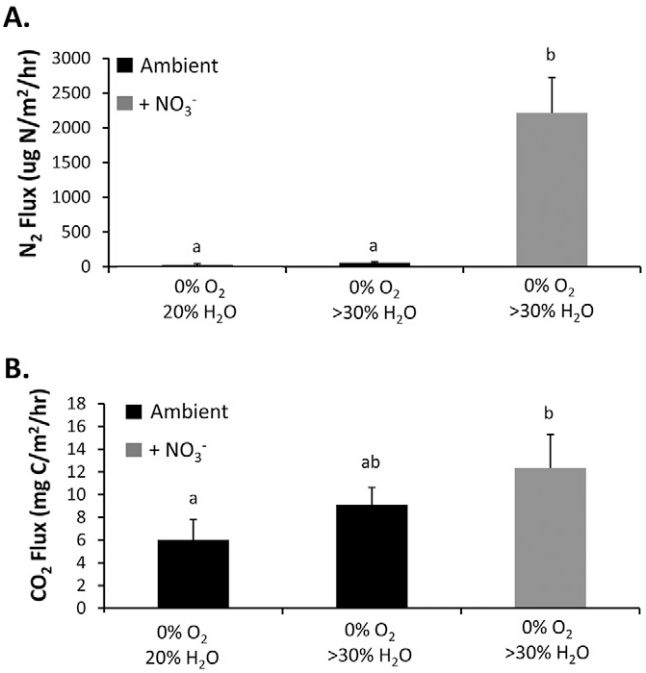

Fig. 3. (A) N_2 and (B) CO_2 fluxes from intact lawn soil cores at 0% oxygen, University of Maryland Baltimore County. Bars represent soil cores under ambient conditions before rainfall (approximately 20% volumetric soil moisture), ambient conditions after rainfall (>30% volumetric soil moisture), and injected with an NO_3^- solution after rainfall. Different letters indicate significant differences across treatments ($p < 0.05$, $n = 4$).

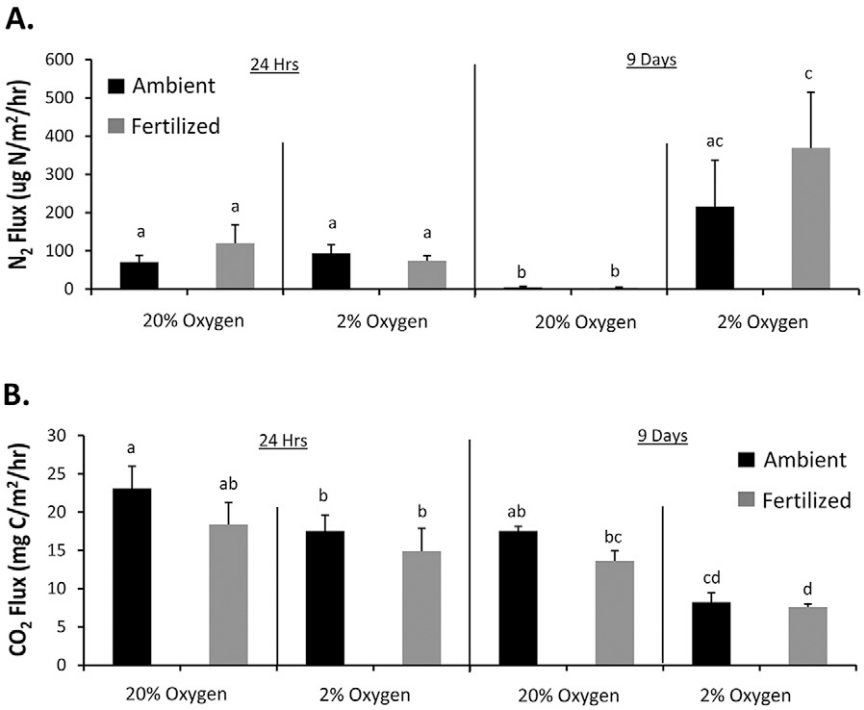

Fig. 4. (A) N_2 and (B) CO_2 fluxes at 20 and 2% oxygen for lawn soil cores collected 24 h and 9 d after a portion of each plot was fertilized in the field (Fertilized) or left untreated (Ambient), University of Maryland Baltimore County. Volumetric soil moisture was approximately 18 to 20%. Different letters indicate significant differences across all treatments and time periods ($p < 0.05$, $n = 4$).

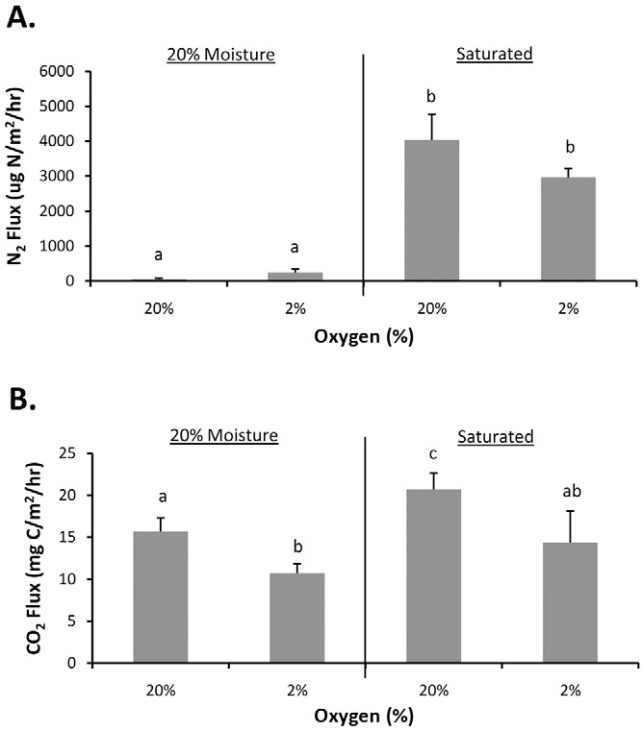

Fig. 5. (A) N_2 and (B) CO_2 fluxes at 20 and 2% oxygen for lawn soil cores collected 9 d after fertilization in the field, then saturated with water in the laboratory, University of Maryland Baltimore County. Different letters indicate significant differences across treatments ($p < 0.05$, $n = 8$).

Discussion

We hypothesized that (i) soil O_2, moisture, and available NO_3^- were the most important controls on denitrification in residential lawns; (ii) denitrification would vary temporally based on these controls; and (iii) N_2 fluxes would be low most of the time, with large pulses following fertilizer addition and precipitation events. While our results generally confirm these hypotheses, the threshold levels of soil O_2, moisture, and NO_3^- availability required to see significant rates of denitrification in lawns (relative to N inputs) were greater than expected. For instance, high rates of denitrification were possible when bulk soil O_2 concentrations were high, and for all oxygen concentrations, lawn soils had to be nearly saturated and recently fertilized to see significant N_2 fluxes. Soil oxygen concentrations were less dynamic (Fig. 1A and 1B), and N_2O fluxes even more dynamic than expected (Fig. 7).

Soil moisture has been linked to soil O_2 in a number of studies (Sexstone et al., 1985; Burgin et al., 2010), so we expected to measure significant drops in soil O_2 concentrations following large storm events; however, this was not the case. Soil O_2 concentrations (measured hourly by three diffusion-head O_2 sensors at each site) never dropped much below ambient atmospheric concentrations over our study period (Fig. 1A and 1B), which included several major storm events. One storm system brought >10 cm of rainfall over a 3-d period (NCDC, 2009), yet soil O_2 concentrations did not change significantly at the sites. These findings suggest that the soils were well aerated and well drained. A lack of redoximorphic features in these soils further supports this conclusion.

We expected denitrification rates to be low when soil O_2 concentrations were high, but this was not always the case. Under wet or saturated conditions, soils amended with fertilizer or NO_3^- had high N_2 fluxes, even when soil O_2 concentrations were at 20% (4039 ± 723 μg N m^{-2} h^{-1}; Fig. 5A). The Apogee sensors likely reflect O_2 levels in macropores, while denitrification likely occurs in anaerobic microsites associated with soil aggregates or particulate organic matter (Parkin, 1987). The fact that soil moisture appeared to be a stronger controller of denitrification rate than soil O_2 thus suggests that factors that influence the flux of O_2 from macropores to denitrifying microsites are critical controllers of denitrification rate. Given that these factors vary from site to site with soil texture, structure, compaction, and other variables, it is likely that relationships between soil O_2 and denitrification rates will vary in both time and space across the landscape. Another key factor affecting the levels of O_2 in soil microsites is bacterial and plant respiration. Lawn soils, which tend to have high soil organic matter, microbial biomass, and labile C and N, have correspondingly high rates of respiration (Qian and Follett, 2002; Raciti et al., 2008; Qian et al., 2010; Fig. 1B and 6B). Since respiration is the primary oxygen-consuming process in soils, we might expect anoxic microsites to form readily in lawns, especially under warm, moist conditions that favor high soil respiration.

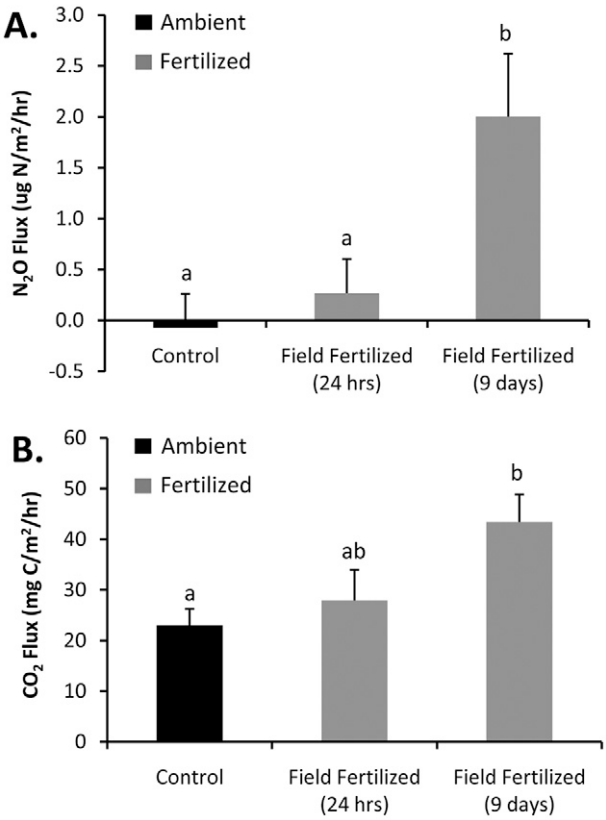

Fig. 6. (A) N_2O and (B) CO_2 fluxes from in situ gas sampling chambers in each of the lawn plots, University of Maryland Baltimore County. Different letters indicate significant differences across treatments ($p < 0.05$, $n = 6$).

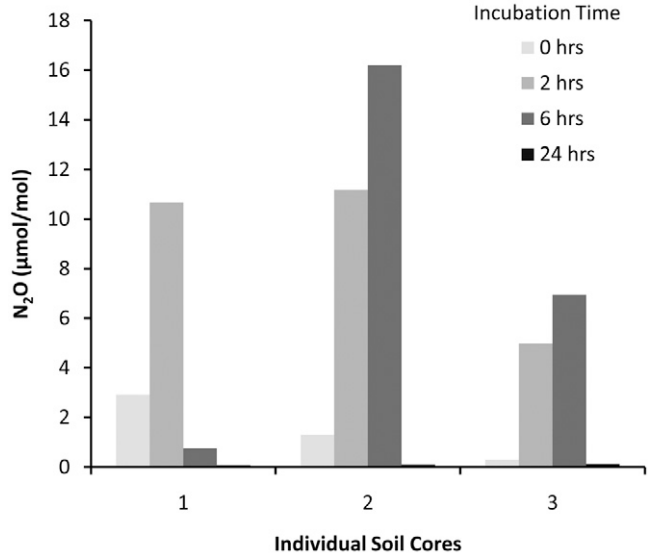

Fig. 7. Laboratory-measured N_2O concentrations in the headspace of individual soil sampling chambers at 0, 2, 6, and 24 h after the start of incubations for fertilized, saturated soil cores, University of Maryland Baltimore County.

The N_2O fluxes we measured in the field suggest that lawns are not a major source of N_2O emissions in our study system. When we fertilized part of the UMBC1 site, we expected to see an immediate and rapid increase in N_2O emissions (one of the intermediary gases in the denitrification process), but N_2O fluxes remained low after 24 h (Fig. 6A). We saw significantly higher N_2O fluxes 9 d after fertilizer application. These enhanced fluxes of N_2O may be the result of greater N availability 9 d after fertilizer application due to the use of a slow-release, sulfur-coated urea product (per standard management practices at the site). The extra time (9 d), combined with a light rainfall 3 d after fertilizer application, may have increased N availability in the soil. However, it should be stressed that even the enhanced N_2O fluxes (<0.2 ng N cm^{-2} h^{-1}) were lower than the mean rates measured in lawn and forest plots at BES (Groffman et al., 2009), which calls into question whether the low but significantly higher N_2O fluxes are a result of the experimental treatment or merely natural variation in the system.

Results from the N-FARM incubations suggest that soil moisture, NO_3^- availability, and the balance of N_2O production and consumption are key factors controlling N_2O emissions from lawns. Fluxes from wet, fertilizer-amended soils support the hypothesis that high soil moisture and NO_3^- availability can lead to high rates of N_2O production (Groffman et al., 2009), but these results also reveal a more complicated relationship between N_2O production and consumption. In fertilized cores, we sometimes saw extraordinarily high concentrations of N_2O followed by very low concentrations (near the detection limit of our instrument) at the next sampling interval (Fig. 7). These findings indicate a lag time between accelerated N_2O production and counterbalancing increases in N_2O consumption, which may have important implications for N_2O fluxes from fertilized soils. Nitrous oxide is a by-product of both nitrification and denitrification, and its solubility and diffusivity control fluxes between the soil and the atmosphere (Heincke and Kaupenjohann, 1999). In the closed incubation system, N_2O could not escape and was generally consumed during the soil incubations. While soil saturation would inhibit N_2O diffusion to a significant degree (Heincke and Kaupenjohann, 1999), it is difficult to say how much N_2O would have been lost to the atmosphere. While long-term, in situ N_2O fluxes from BES lawns suggest that they are not major sources of N_2O emissions, these measurements are not keyed to fertilizer or rainfall events and may miss periods of high N_2O emissions (Groffman et al., 2009). Thus, like studies in the western United States (Kaye et al., 2004; Bijoor et al., 2008; Hall et al., 2008; Townsend-Small and Czimczik, 2010; Townsend-Small et al., 2011), we cannot yet conclude that lawns are an insignificant source of N_2O in the study area.

Closing the Nitrogen Budget

Using a combination of laboratory and field measurements, we calculated an annual denitrification rate of 14.0 ± 3.6 kg N ha^{-1} yr^{-1} for the lawns in this study (see the Calculations and Statistics section and Table 1). This calculation is based on long-term field and laboratory measurements of soil moisture and oxygen conditions, but does not account for other factors that might influence N_2 fluxes, such as temperature, seasonality (soil cores were collected during the spring), plant species, soil type, and climate variation. Further, while the sum of N inputs (109.2 kg N ha^{-1} yr^{-1}) and losses or sequestration (111.1 kg N ha^{-1} yr^{-1}) in our mass balance (Table 2) yields a nearly balanced N budget, there is significant uncertainty surrounding some of the numbers in the budget. The term with

Table 2. Nitrogen mass balance calculated for study lawns, University of Maryland Baltimore County.

Nitrogen flux	Residential	Forest	Source
N inputs			
Atmospheric deposition (kg N ha^{-1} yr^{-1})	11.2	11.2	Groffman et al., 2004
Fertilizer (kg N ha^{-1} yr^{-1})	98.0	0	This study (UMBC1 lawn plot)
Hydrologic N losses			
Leaching (kg N ha^{-1} yr^{-1})	14.1	4.4	Groffman et al., 2009
Gaseous N losses			
N_2O (kg N ha^{-1} yr^{-1})	<1	<1	Groffman et al., 2009
Denitrification as N_2 (kg N ha^{-1} yr^{-1})	14.0	N/A	This study combined with long-term soil moisture records at Baltimore Ecosystem Study (http://beslter.org).
N sequestration			
N accumulation in soils (kg N ha^{-1} yr^{-1})	83.0	N/A	Residential sites with agricultural land-use history (Raciti et al., 2011a).

the largest uncertainly is soil N sequestration in residential lawn soils, which was estimated using a chronosequence of lawns in the study area (Raciti et al., 2011a) and may not be an accurate estimate for an individual lawn. Atmospheric N inputs may also be somewhat higher or lower than reported in our budget, since they reflect estimates from a USEPA Clean Air Status and Trends Network (CASTNET) site that is approximately 30 km to the south. Finally, hydrologic N losses are based on the long-term average for the site, but there is interannual variation in this rate (Groffman et al., 2009).

Despite these significant uncertainties, our calculations suggest that denitrification is an important part of the N budget for residential lawns (Table 2). The N_2 flux we calculated (14.0 ± 3.6 kg N ha^{-1} yr^{-1}) was comparable in magnitude to atmospheric N deposition (11.2 kg N ha^{-1} yr^{-1}), NO_3^- leaching losses (14.1 kg N ha^{-1} yr^{-1}), and was equal to approximately 15% of mean fertilizer N inputs to home lawns in the study area (83.5 kg N ha^{-1} yr^{-1}; Law et al., 2004). This rate is within the range of denitrification estimates obtained by Horgan et al. (2002a,b) using a ^{15}N mass balance approach, though a significant fraction of the ^{15}N tracer used in those studies could not be accounted for at the end of the experiments. Our annual flux calculation indicates that relatively short periods of saturation may be responsible for the majority of denitrification in our well-drained lawn soils. Lawns with limited drainage (or at low points in the landscape) could be hotspots for denitrification in residential areas. Our results suggest that denitrification is an important means of removing reactive N from the residential landscape, but varies markedly in space, time, and with factors that affect soil saturation (texture, structure, compaction) and NO_3^- availability (fertilization). While the annual denitrification rate we calculated was substantial, there is evidence that soil organic matter may be an even larger N sink in residential landscapes, particularly for sites with a history of agricultural land use before development (see Raciti et al., 2011a). Further work is required before the findings of this, or the aforementioned study, can be generalized to a wider range of residential lawns and soils.

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